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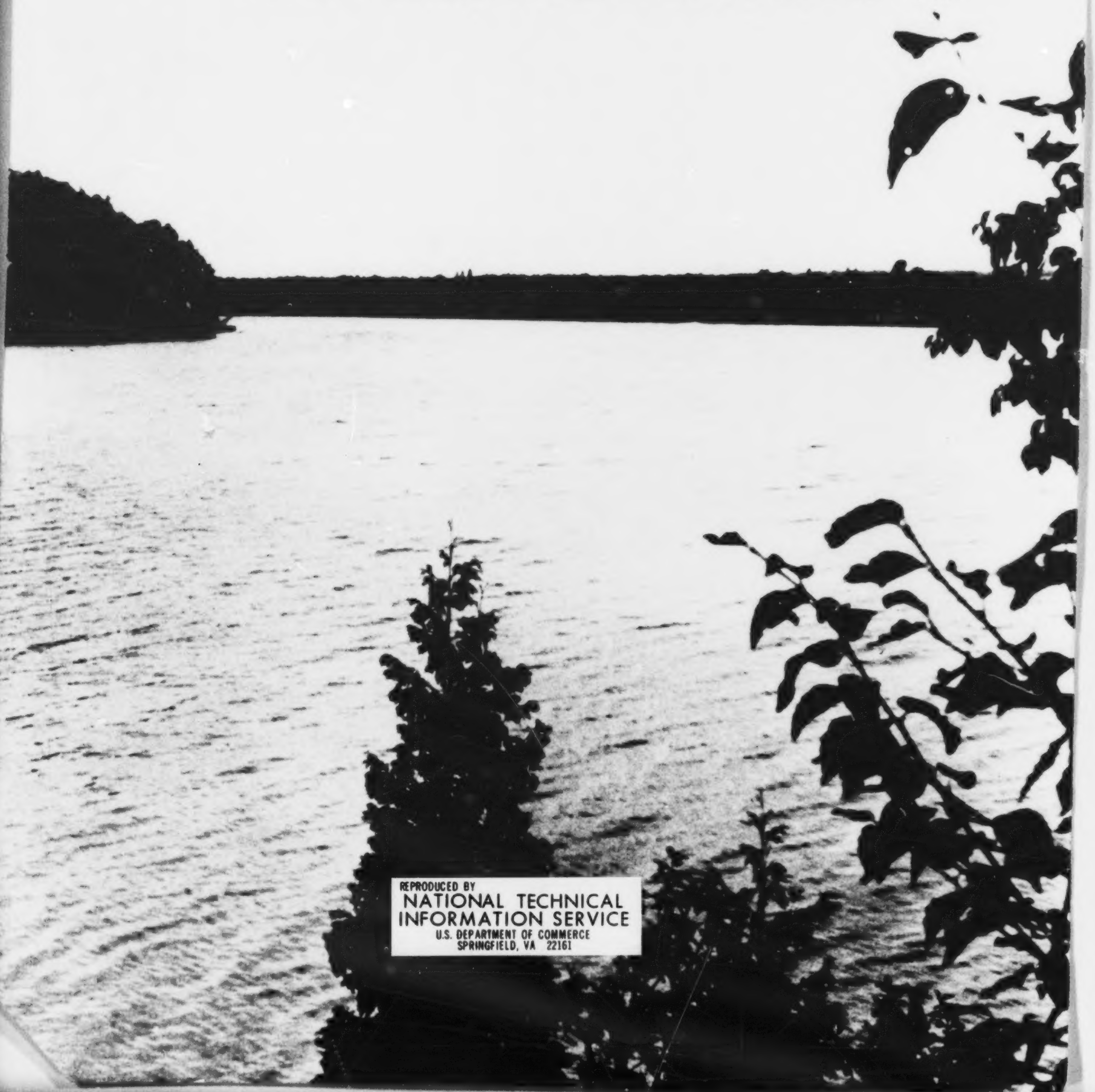

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Section 304(j) of the 1977 Amendments to the Clean Water Act requires the U.S. Environmental Protection Agency (EPA) to publish a biennial report on "methods, procedures, and processes as may be appropriate to restore and enhance the quality of the Nation's publicly-owned freshwater lakes." To fulfill that legislative requirement EPA joined with the Organization for Economic and Cooperative Development (OECD), which just completed a decade-long study of eutrophication, to sponsor an International Symposium on Inland Waters and Lake Restoration.

Scientists and project managers involved with freshwater lakes projects throughout the world presented the results of their investigations to this Symposium. Ninety-one presentations are published in these proceedings.

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RESTORATION OF LAKES AND INLAND WATERS

International Symposium on Inland Waters and Lake Restoration

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September 8-12, 1980
Portland, Maine

U.S.ENVIRONMENTAL PROTECTION AGENCY
OFFICE OF WATER REGULATIONS AND STANDARDS
WASHINGTON, D. C.

December 1980

FOREWORD

This second biennial report on the protection and restoration of our Nation's freshwater lakes comes at a particularly propitious time. Not only our own country, but other nations are beginning to demonstrate significant progress in meeting the environmental challenges to restore and protect freshwater lakes.

Section 304(j) of the 1977 Amendments to the Clean Water Act requires the U.S. Environmental Protection Agency (EPA) to publish a biennial report on "methods, procedures, and processes as may be appropriate to restore and enhance the quality of the Nation's publicly-owned freshwater lakes." To fulfill that legislative requirement EPA joined with the Organization for Economic and Cooperative Development (OECD), which just completed a decade-long study of eutrophication, to sponsor an International Symposium on Inland Waters and Lake Restoration.

Scientists and project managers involved with freshwater lakes projects throughout the world presented the results of their investigations to this Symposium. Ninety-one presentations are published in these proceedings.

Seven hundred fifty (750) people attended from 35 countries and 46 States to hear and discuss the state-of-the-art. This demonstrates the intense interest in lake restoration. Two years ago, 450 attended the conference in Minneapolis.

We are learning to understand what creates problems in our lakes. Lake restoration and protection is a developing science. It is a challenge both to seasoned scientists who have worked with environmental problems for many years and to their younger counterparts. Both came to this Symposium, and shared the podium to explain innovative investigations that have produced methods and procedures that are working.

Freshwater lakes are being protected and restored in the United States and throughout the world. We expect the momentum behind this effort to grow stronger and become even more effective over the next few years, as the States take over the responsibility for the restoration and protection of their publicly-owned freshwater lakes from the Federal government.

Steven Schatzow

Steven Schatzow
Deputy Assistant Administrator
Office of Water Regulations and
Standards

INTRODUCTION

The U.S. Environmental Protection Agency's Clean Lakes Program has demonstrated that principal causes of our lake problem can be identified, and comprehensive, cost-effective solutions can be developed and successfully implemented for most of our lake problems.

The symptoms of lake eutrophication are obvious to the public. Aquatic weeds interfere with swimming and boating. Observers notice that the once clear water has become increasingly dark and murky. Fishermen rely on their memories to describe the excitement of the sport fishery now diminished.

Eutrophication is not the only threat to our enjoyment and use of our Nation's lakes. Heightened public awareness of the dangers and widespread occurrence of toxic pollutants has stimulated concern for the presence of toxics in lakes. Heavy metals and synthetic organic compounds resulting from expanding urbanization and industrialization may be interfering with the natural systems of our lakes. Toxics impact the ability of lake users to enjoy lake recreational activities and the quality of their drinking water and fish harvested from the lakes.

Numerous techniques have been developed to reduce the availability of nutrients and slow down the eutrophication process. Simply treating the problems, however, is not the most effective solution. The Clean Lakes Program has demonstrated that controlling the cause of the problem is the best approach. Point source and nonpoint source loadings in the lake watershed must be reduced to an acceptable level to achieve the desired improvement in lake water quality.

In many instances, nonpoint source control is the most important aspect of lake restoration. The complexity and extensiveness of nonpoint sources make their control difficult. In certain lake watersheds, thousands of acres of forests, agricultural land, and urbanized areas must be studied and subjected to effective management strategies. Public awareness of the connection between watershed activities and lake water quality is essential.

Lake watershed management strategies have used contour plowing, manure handling systems, timber harvesting practices, street sweeping, and stormwater and sedimentation basins to successfully prevent these nonpoint source pollutants from entering the lake.

The science of lake protection and restoration has progressed over the past few years. It is essential to continue these efforts to improve our knowledge of problems and our abilities to solve them. The future of our lakes is bright, provided we continue to:

- Improve this Nation's capability to predict which pollutants have the greatest impact on our lakes so that lakes can be preserved and restored in the most cost-effective way.
- Improve the effectiveness of pollution control and treatment methods, while at the same time, reduce their costs and use of energy.
- Find constructive uses for water materials such as those dredged from lake bottoms and the byproducts of farming (manure) and industry (toxics).
- Improve our understanding of the eutrophication process and nutrient recycling within lakes—mechanisms, importance, and impact on the lake systems.
- Evaluate lake restoration to improve the technology, quality, and duration of lake pollution control.
- Develop innovative pollution control and treatment technology addressing toxics.

Learning from others' experience is a basic truism of human existence, and it certainly applies to the rapidly growing science of limnology. Lakes throughout the world share similar problems. Various techniques to solve lake problems, whether developed from farm ponds in Australia or reservoirs in Germany, may be applicable to similar situations wherever they exist.

Not only scientists, but laymen must learn the fundamentals of lake ecosystems and their protection and restoration. Lakes' problems are not solely the province of limnologists, nor of governmental officials, but of the people who use and are affected by those lakes. A Clean Lakes Program is by its very nature a "grassroots program." That's what makes the Clean Lakes Program work.

Government will continue to play an essential role in restoring lakes, but it will be Government closest to those whose lakes need help. State and local Governments are best able to set their own priorities for the restoration and/or protection of lakes and integrate lakes into their total water quality management program.

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WELCOME

LESLIE A. CAROTHERS
Deputy Regional Administrator
Region I, U.S. Environmental Protection Agency
Boston, Massachusetts

On behalf of EPA's New England regional office, I want to welcome you to New England and the International Symposium on Lake Restoration. I am standing in today for Bill Adams who is vacationing in our western national parks. In fact, his itinerary for today places him at the Great Salt Lake, and I know he sends his greetings to all of you. The Clean Lakes Program in close to his heart. When Bill Adams served as commissioner of the Maine Department of Environmental Protection before coming to EPA, he initiated several of the first federally supported lake restoration projects. The State of Maine continues to be a pacesetter in lakes protection under Henry Warren's aggressive leadership.

We are proud of the success of the Clean Lakes Program throughout New England. To date, we have approximately 40 operating Clean Lake projects totaling approximately $10 million. Early results of these projects are encouraging. Recreational uses have been partially restored to Morses Pond and Nutting Lake in Massachusetts, Lake Bomoseen in Vermont, and Annabessacook Lake in Maine. I understand that you will be learning more details of these projects during this conference.

We are particularly pleased to have recently awarded the first lake protection grant to the Cobbossee watershed district in Maine. (One of the things you learn at this conference is how to pronounce Maine's Indian names!) This grant will provide financial assistance for remedial and preventive activities to protect 15 lakes and ponds in the Cobbossee watershed in central Maine. A previous planning study indicated that over 70 percent of the annual nutrient loading to these lakes comes from both nonpoint agricultural runoff and stormwater runoff. The impact of the projected land use changes and population growth will result in phosphorus loading increases sufficient to trigger nuisance algae blooms in 11 of 15 lakes in the district.

With the positive action to improve land use management, we expect that the existing high quality lakes in the area can be maintained. We are encouraged to see other areas and States pursuing similar programs.

In addition, Region I has taken every opportunity to use other EPA programs to benefit our recreational lakes. These include Federal grants for construction of municipal waste treatment facilities and the Rural Clean Water Program. The restoration of Lake Winnisquam in New Hampshire and Rangely Lake in Maine are just two examples of the successful use of the construction grant program to reduce pollution adversely affecting important lake resources.

The RCWP is a multi-agency water quality improvement program directed at abating pollution from farming practices. It provides for planning and implementation funds to correct activities which are adversely affecting stream and lake quality. In Region I, the St. Alban's Bay watershed in Vermont has been selected for FY 1980 funding. The combined efforts of the ongoing municipal construction program for the City of St. Alban's and the Rural Clean Water Program, tackling the basin's nonpoint source pollution, will greatly clean up St. Alban's Bay and restore long impaired recreational uses in Lake Champlain.

I see from the conference agenda that you are beginning four days of intensive review of the latest scientific research in your field. Although your work is complex and difficult, I think you are fortunate because the results of your efforts provide benefits that are seen and appreciated by people who love our lakes, even if they do not know what limnology means. It must surely be rewarding to help to preserve and enhance resources of economic and recreational value and places of beauty and peace for our people to enjoy. I wish all of you well in your important work.

THE U.S. EPA CLEAN LAKES PROGRAM

STEVE SCHATZOW
Deputy Assistant Administrator
Office of Water Regulations and Standards
U.S. Environmental Protection Agency
Washington, D.C.

The enthusiasm for this symposium mirrors the enthusiasm we see in this country for the Clean Lakes Program. Granted, it is one of the few EPA programs that is non-regulatory in nature. We are not forcing people to clean up their lakes — we are helping them. They are even digging into their own pockets and using their own muscles to make the clean lakes projects work for them.

And, as a result, the Clean Lakes Program is doing more than protecting our lakes' resources — it is rekindling the grass roots involvement that is the key to this country's political system. Our citizens are solving their lakes problems at the local level: we at the Federal level are helping — and together, we're doing something no other Federal program is doing — we're turning a profit!

A recent study showed that every Federal dollar invested in clean lakes projects is returning $8.30 in benefits! That's something to brag about!

And it is but one indication of the success our Clean Lakes Program is achieving. As they say in our television commercials, we have come a long way in the 8 years since the senior Senator for Minnesota — you know him now as Vice President Mondale — collaborated with Senator Quentin Burdick and Congressman Donald Fraser to maneuver the Clean Lakes Act through Congress. Together, they laid the cornerstone of the Clean Lakes Program.

In that law, Congress declared that our Nation's publicly owned freshwater lakes should be protected and restored. It gave that job to the Environmental Protection Agency. We took the responsibility very seriously because we were concerned with a number of problems we saw with that mandate.

In the first place, how could we start a national program when we knew so little about the size of the problem? We still don't know for sure how many lakes we have — there's an argument about that number every time we publish a book!

But there was another problem — we believed that other pollution controls — the permits limiting pollution discharge, for example — would be enough to protect lake quality. And we also questioned whether technology was adequate to deal with lake eutrophication problems. Finally, we were very concerned with the cost-effectiveness of lake restorative measures.

So we proceeded deliberately, working out those initial doubts, using our first $4 million appropriation in 1975 to initiate pilot lakes projects.

Today, we have no doubts, only proof that a national Clean Lakes Program is needed and is working. We believe that about 10,000 of our publicly owned lakes need pollution control and restoration — that's a sizable national problem that would cost us nearly $5 billion.

We have also discovered that those other pollution control programs cannot alone solve the lakes' problems. Granting permits for discharge helps, but nonpoint source pollution and watershed management are just as important. We have also discovered that the technology to restore and protect lakes not only exists but is rapidly becoming more sophisticated — you are going to spend this entire week discussing that! And as for cost-effectiveness, we've already told you that the program in its short lifetime is returning benefits of more than 8 to 1.

So our initial task of discovering and testing that technology has proved successful. Where do we go from here?

I just quoted you a $5 billion figure to clean up those 10,000 lakes we believe need help. Realistically, we can't afford that — our Nation's financial pie just doesn't cut into that big a slice for cleaning up our lakes. What do we do? We look at the problem realistically and we come up with a solution. A realistic solution that we call our 5-year strategy. A solution that makes the most of our limited funds to benefit the most people.

We have set a goal to protect or restore at least one lake with water quality suitable for contact recreation within 25 miles of every major population center. This goal will take at least $150 million in Federal funds but it will serve almost all of our population. Frankly, our current rate of budget support will not enable us to meet the goal in 5 years. We may have to stretch it out somewhat.

The goal, though, is within our reach. One of the facts we have learned in these first years of the program is that 99 percent of us live within 50 miles of a publicly owned freshwater lake. And one-third of us live 5 miles or less from a lake. So our goal is right on target, particularly today when the price of gasoline is soaring — Americans want recreation close to home.

To meet this goal, our strategy has five objectives that must be considered when any clean lakes project is approved. The first is that the project must maximize public and environmental benefits. It must also coordinate with other Federal and State programs. It must emphasize pollution controls, particularly for nonpoint sources before it uses any in-lake devices. The project must also emphasize the Federal-State partnership: and, as a final objective, the State must continually evaluate the program to maintain high quality pollution control practices.

We have developed a technical guidance manual for State use and a citizen's guide on how to use the Clean Lakes Program. Both will be available within the next few weeks. The manual can be reviewed at the EPA exhibit here.

These publications and our 5-year strategy all communicate two overall objectives of our Clean Lakes Program. In all our projects, we encourage best management practices and pollution control in lake watersheds. We will not make an award unless pollution controls required by the Clean Water Act are in place or are progressing — and we also require control of nonpoint sources of pollution according to procedures developed under the Act.

Our second overall objective stems from the first. We insist on program integration. You may remember I mentioned that as one of the strategy objectives. Many Federal, State, and local programs have some impact on water quality.

If used wisely, a combination of these programs can meet a common goal at lower cost, and certainly avoid duplication of effort. We have worked with the Heritage Conservation and Recreation Service of the Department of the Interior to restore 59th Street Pond in New York's Central Park — and a current project at Broadway Lake in South Carolina is using resources from the Department of Agriculture and a State agency to supplement the clean lakes award.

But the most essential element of the Clean Lakes Program is still citizen participation. To date we have spent nearly $60 million for over 200 projects in 46 States. But these are not just government handouts. These are matching funds — somebody besides the Federal Government must come up with 50 percent of the cost of a lakes project. That could be the State. More often, it is the local government because it was the lakeside community that first saw those weeds choking their waterway, and smelled the rotting fish, and decided to do something about it.

So once their project is approved, they come up with the money. Only sometimes it isn't money. Remember I mentioned muscle earlier in this speech? The people of Scotia, N. Y. rounded up privately owned tow trucks, hooked them to tree trunks that had to be removed from their lake, and pushed while the trucks pulled. That's really working for your lake!

And that's probably why the Clean Lakes Program has been so successful in this country. The public can actually see and smell their problem and they can do something about it. Once they've put their money and muscles into it, they're careful to maintain their lakes, and they look around to see how else they can improve their communities. One town built parks: another rebuilt decaying neighborhoods.

Congress just thought they put the responsibilty for protecting and restoring lakes on the Environmental Protection Agency.

They really put it on the people of this country, and Americans have accepted the challenge to a point that EPA would not have dreamed possible 8 years ago. We are truly proud of our program.

And now let me turn to the more immediate goals of this symposium. We have an impressive array of experts on the program who will provide us with the most up to date information on the science of lake restoration. I want to extend an especially warm welcome to our colleagues from the many countries outside of our borders who are participating in this symposium. We are especially appreciative of the cooperation of the Organization for Economic Cooperation and Development in sponsoring the symposium.

The program is impressive. The topics to be presented indicate we do know a lot about the causes of lake eutrophication and the techniques of restoration.

Too many times scientists are too modest. This conference is going to give us the opportunity to spread their knowledge across the world.

We are rather proud in this country of the progress made in a few short years. I discussed some of the results of our national program a few minutes ago. We know that lake restoration techniques work. You will hear about them in more detail during the week. Our technical efforts have been increasingly focused on watershed control technology. Since many of the lakes we want to restore are in or near cities, urban runoff control technology is high on our list of the problems which demand attention. We look forward to this symposium to provide some of the answers.

We join with OECD to host you at this symposium. I believe this is a great opportunity to exchange information on lake restoration that may not present itself for years to come. The discussions that take place here can provide a springboard for a worldwide restoration of our lake resources during the next decade.

*Mr. Schatzow's comments which suggest a Federal financial commitment in future years do not reflect the current Federal position. Decisions made in the FY 1982 budget resulted in the elimination of Clean Lakes Program funds due to higher environmental priorities. It is anticipated that local communities and States will assume full responsibility for lake cleanup in an appreciable number of projects.

LOCAL COMMITMENT TO LAKE RESTORATION: THE COBBOSSEE WATERSHED EXAMPLE

THOMAS U. GORDON
Executive Director
Cobbossee Watershed District
Winthrop, Maine

ABSTRACT

Successful lake management requires a strong local commitment to lake restoration and protection. The Cobbossee Watershed District has integrated Federal, State, local, and private resources to restore eutrophic lakes. The Cobbossee Watershed District is the State of Maine's only local unit of government devoted exclusively to lake management. The District covers a watershed of 622 square kilometers, including four eutrophic and 11 mesotrophic lakes of 30 to 2,259 hectare in size. The District's primary objective has been the restoration of 575-hectare Annabessacook Lake. Utilizing a Clean Lakes grant, the Cobbossee Watershed District established a cost-sharing program to construct agricultural waste management facilities in the Annabessacook, Cobbossee, and Pleasant Pond drainages. A hypolimnetic application of aluminum sulfate and sodium aluminate also was used on Annabessacook Lake for nutrient inactivation. Completion of both phases of the project required coordination of Federal and State agencies, as well as agricultural groups and lakeshore property owners' associations. Follow-up monitoring of Annabessacook Lake has found significant reductions in phosphorus and chlorophyll, and improvements in visibility. Similar responses are anticipated in the two downstream lakes.

INTRODUCTION

Water pollution control has long been a function of government, with specific responsibilities divided among the national, State, and local levels. The Federal Water Pollution Control Act Amendments of 1972 provided the first national initiative for lakes restoration through section 314 of the law, the Clean Lakes Program. Prior to 1972, many States, most notably those in the Great Lakes Basin, had developed their own programs for protecting and improving their publicly owned lakes and ponds.

Today, State agencies have the primary responsibility under the Federal Clean Water Act for diagnosing and treating lake water quality problems. New EPA regulations, which channel all lake restoration grants through the State water pollution agencies, have reinforced the States' central position in the Clean Lakes Program.

Lake management activities have also been conducted at the sub-state level by a variety of public agencies, citizens' groups, and private enterprise. The diversity of American local government — cities, counties, towns, special districts — greatly complicates any summation of local roles in lake restoration. Generally, successful implementation of restoration projects and protection strategies for most lakes will require a strong local concern for water quality. Furthermore, effective lake management efforts will require careful integration of limited Federal, State, local, and private resources. The Cobbossee Watershed District has provided one example of local leadership in lake restoration.

DESCRIPTION OF THE COBBOSSEE WATERSHED

The Cobbossee Watershed, a sub-basin of the Kennebec River Basin, is located approximately 80 kilometers north of Portland, Maine. The 622 square kilometer watershed consists of a chain of 24 lakes and ponds, ranging in size from 30 to 2,259 hectares. Four of these lakes — Annabessacook, Cobbossee, Little Cobbossee, and Pleasant — are culturally eutrophic. Glacial till predominates in the surficial geology of the watershed.

Approximately 25,000 people reside in the watershed on a permanent basis. During the summer months, the population increases by 60 percent, reflecting the significant tourist and second-home economy of the lakes region. The lakes also serve as a recreational resource for 47,000 people residing in communities peripheral to the watershed. The watershed remains predominantly forested (approximately 75 percent of the land area), with agricultural (12 percent) and residential (8 percent) land uses also significant.

HISTORY OF WATER QUALITY PROBLEMS

Historically, water quality concerns in the Cobbossee Watershed have focused on Annabessacook Lake. For more than 150 years, the tributaries of Annabessacook Lake served as conduits for municipal and industrial effluent. The earliest reports of serious water quality

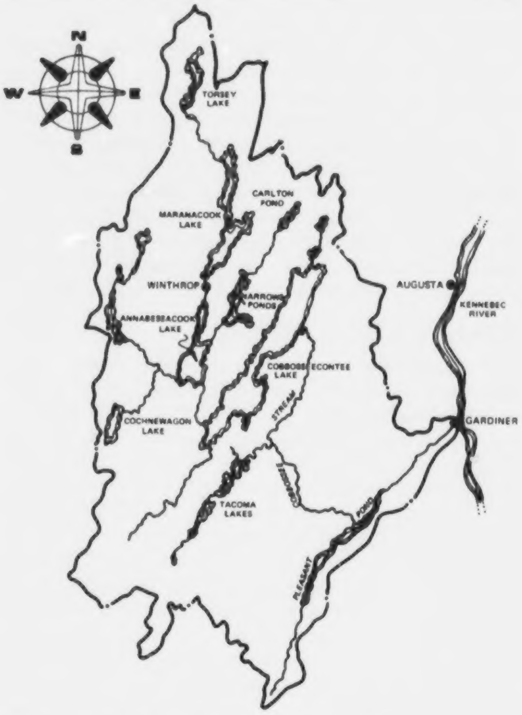

Figure 1 — The Cobbossee Stream watershed.

degradation occurred in the late 1930's. Despite years of public complaints, no intensive evaluation of the problem was made until the mid-1960's (Smith and DeWick, 1965). A study of the lake by the Maine Water Improvement Commission found violations of water quality classifications and standards. Late summer Secchi disk visibilities ranged from 0.76 to 1.83 meters, caused by intense blooms of *Anabaena* and *Aphanizomenon*. The annual phosphorus loading to the lake from municipal and industrial sources was estimated at 14,000 kilograms. Furthermore, Cobbossee Lake, located immediately downstream from Annabessacook and once famous for its salmon and other fisheries, also experienced a significant decline in water quality.

As a result of these findings, an 18-kilometer trunkline sewer was constructed to divert all point sources from the Annabessacook watershed to treatment facilities on the Kennebec River in the city of Augusta. Thus, wastewater discharges from the town of Winthrop terminated in 1972; the remaining discharges from the town of Monmouth continued through 1976, when a 10-kilometer extension of the trunkline sewer was completed. Elimination of these discharges reduced phosphorus loading to Annabessacook Lake by approximately 90 percent (Sage and Moran, 1977). However, the lake's ambient concentrations of total phosphorus remained above 15 parts per billion, and nuisance algal blooms persisted.

Despite the efforts being made to control nutrient inputs in the early 1970's, there was a strong public perception that not enough was being done to restore Annabessacook and Cobbossee Lakes and to protect the other lakes in the watershed from similar problems. From 1964 through 1972 lakeshore property owners on Annabessacook Lake took their own remedial action by treating the lake with algacides. Over 8 years, approximately 30 tons of copper sulfate were applied, with diminishing effectiveness as copper-resistant types of algae began to predominate. A proposed sodium arsenate application was prohibited by State health officials. Two small-scale attempts to aerate the lake failed to produce any noticeable change in water quality.

Frustrated by 30 years of failure in improving lake water quality, lakeshore property owners on Annabessacook and Cobbossee began working with municipal officials of the Southern Kennebec Valley Regional Planning Commission to develop a comprehensive strategy for lake restoration in the 1970's. Given the highly experimental nature of lake restoration technology (Imhoff, 1971), efforts concentrated on creation of an institution to develop and implement appropriate restoration techniques. The Federal and State Governments were perceived as the primary sources for funding the necessary research on the lakes. Since the lakes and streams of the Cobbossee Watershed fall under the jurisdiction of as many as 16 separate municipalities, 2 counties, 4 water districts, and 5 sanitary districts, no one unit of local government could

Table 1. —

	Annabessacook	Cobbossee	Pleasant Pd:
Morphometry			
Surface area	575 ha.	2,244 ha.	237 ha.
Mean depth	5.3 m	8.07 m	2.68 m
Maximum depth	14.9 m	30.48 m	7.9 m
Total drainage area	85 mi^2	131.4 mi^2	217 mi^2
Direct drainage	21.8 mi^2	46.7 mi^2	23.6 mi^2
Flushes per year	4.5	1.2	5.6
Land Use Characteristics (Direct drainages in percentages)			
Forest and reverting fields	69	65	73
Developed	12	11	8
Agriculture			
active, fields	16	20	16
active, tilled	1	0	1
Other	2	4	2

provide a comprehensive approach to lake management.

Instead, a quasi-municipal, special-purpose district was proposed to assume the tasks of lake research, restoration, and protection for the Cobbossee Watershed. In addition to providing a single jurisdictional unit for watershed planning, the lake district often has independent powers of taxation, the ability to focus all its resources on a single issue, and a clearly defined group of constituents who can support the district's efforts (Gordon, 1977). Lake protection and rehabilitation districts have subsequently found widespread acceptance in Wisconsin and other States.

The Cobbossee Watershed District was authorized by the Maine Legislature in 1971. The District's legislative charter called for establishing a Board of Trustees of up to 17 members, appointed by 10 municipalities and 3 water districts designated as members of the District. To become operational, the District had to be ratified by public referenda in each of the municipalities. In November 1972, 8 of the 10 municipalities voted to join the District. More than 80 percent of the voters supported creation of the District, a particularly significant figure given the uncertainties about the agency's tax assessment potential.

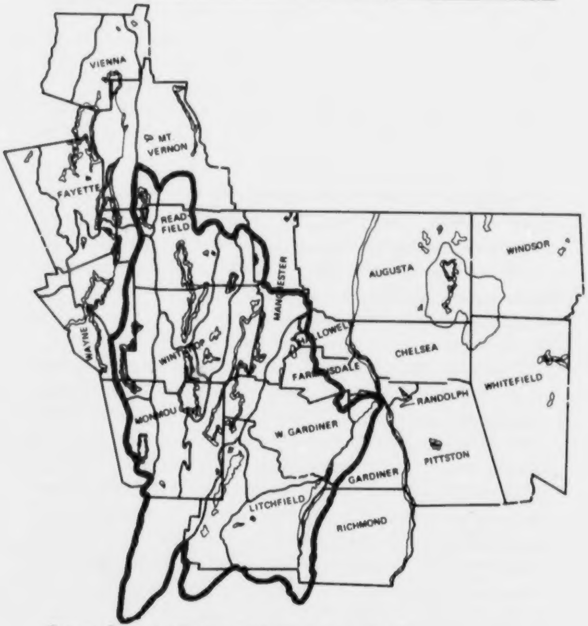

Figure 2. — Lakeshed and town boundaries.

The District's legislated purposes are to protect, improve, and conserve the lakes, ponds, and streams of the Cobbossee Watershed for the public health and welfare and for the benefit of residents and property adjacent to these waters. To do so, the District is authorized to do any and all things necessary to improve water quality. The District is also authorized to own and operate the 22 small dams in the watershed. Specific powers of the District include eminent domain, taxing all property within member municipalities, bonding for major capital expenses, and the authority to pass rules and regulations. To date, the District has not acquired dams or passed new water quality regulations. Instead, the District has emphasized a voluntary, cooperative approach to watershed management problems.

The operating budget for the District is approved annually by registered voters of the member municipalities at a public budget meeting. Once approved by the voters, each municipality must pay a share based on the proportionate value of its shoreland property. The operating budget for 1973 was $25,000, derived totally from local taxes. In 1980, the District's budget is approximately $90,000, with 60 percent of the funds derived from Federal and State grants. The consistent support of local taxpayers has allowed the District to maintain a stable program and permanent staffing.

DIAGNOSTIC STUDIES

The District's initial efforts at water quality management centered on individual subsurface wastewater disposal systems. In 1974 and 1975 a comprehensive sanitary survey was conducted of approximately 1,200 lakeshore residences. Information was gathered on the design, location, age, usage, and maintenance of wastewater disposal systems. Less than 5 percent of the systems surveyed were found to be discharging effluent to the lakes or ground surface. More than 50 percent of the systems, however, were classified as inadequate, according to the upgraded standards of the 1974 Maine State Plumbing Code (Freedman, et al. 1977). At the time of the survey, no definitive relationship between subsurface wastewater disposal and lake water quality had been established. Subsequent literature review and research (Sage and Moran, 1977; Beals, 1980) has led the District away from subsurface wastewater disposal as a significant source of phosphorus loading to its lakes.

Regular water quality monitoring of the Cobbossee Watershed lakes intensified in 1976 with funding of an Areawide Water Quality Management Plan by EPA, pursuant to section 208 of the Federal Water Pollution Control Act Amendments of 1972. As a part of the 208 planning program, the District conducted intensive lake studies on 11 lakes in the area. These studies attempted to define the sources of phosphorus loading to the lakes. In the Annabessacook Lake watershed, phosphorus concentrations and stream flows were monitored on five major tributaries and the lake outlet. In-lake monitoring was also conducted, with particular emphasis on spring and fall overturn. The water quality data were then used in the Dillon-Rigler model to

Table 2. — Phosphorus runoff rates for the Cobbossee Watershed.

Source	Phosphorus Runoff (kg/ha)
Forests	.03 ± .01
Clearcut forests	.30 ± .10
Reverting fields	.03 ± .01
Cultivated crops	1.0 ± .5
Manured fields	1.6 ± .4
Village (storm sewers)	1.1 ± .2
Residential	
nearshore	.9 ± .3
remote	.45 ± .15
Septic systems	0-20% annual input
Precipitation (per ha lake surface)	.1 ± .04[1]

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[1] Annual input = 1.5 kg/cap-yr for permanent residences; .5 kg/cap-yr for seasonal.;

estimate phosphorus loading in the lake. Finally, existing land uses in the lake's watershed were examined. By applying appropriate phosphorus runoff rates to the acreage in each land use category, the significance of various land uses in phosphorus enrichment of the lake could be estimated.

The completed lake studies established priorities for phosphorus loading reductions on Annabessacook Lake, Cobbossee Lake, and Pleasant Pond (see Table 3). The primary watershed source of phosphorus loading to these lakes, and almost all lakes in the District, was found to be agricultural runoff, principally from animal waste spread on frozen or snow-covered ground during the winter. Only one farm of 26 surveyed in the watershed had winter manure storage facilities (Sage, 1977c.). Recycling of phosphorus from bottom sediments in Annabessacook Lake was also a significant source of loading to that lake. These diagnostic studies, funded by the 208 planning program, provided the basis for a Clean Lakes grant application to EPA in March 1977.

Table 3. — Phosphorus loading control priorities.

Annabessacook Lake	Kilograms	Percent
*Lake bottom sediments	1,500	36
*Agricultural runoff	1,000	24
Upstream lakes	1,000	24
Urban runoff	450	11
Forest runoff	100	2
Precipitation	60	1
Septic leachate	30	1
	4,200	
Cobbossee Lake		
*Upstream lakes	4,900	56
*Agricultural runoff	2,600	29
Urban runoff	850	9
Precipitation	225	2
Forest runoff	200	2
Septic leachate	100	1
	8,900	
Pleasant Pond		
*Agricultural runoff	1,500	78
Urban runoff	250	13
Forest runoff	125	7
Precipitation	25	1
Septic leachate	25	1
	1,930	

* - Priority sources for lake restoration.

AGRICULTURAL WASTE MANAGEMENT

Controlling agricultural nonpoint sources of phosphorus in the watershed presented several challenges. Appropriate designs for animal waste management practices had to be developed to meet the varying site conditions, types of animal wastes, and existing farm management practices. Financing costly waste management facilities required establishing a cost-sharing program, using EPA Clean Lakes funds. Finally, working relationships with existing agricultural service agencies had to be defined.

Effective containment of animal waste for the duration of Maine's winters requires storage capacity for at least 6 months. This can reduce phosphorus runoff by as much as 70 percent (Porter, 1975), thereby minimizing water quality impacts and conserving fertilizer for food production during the summer months. The typical facility for daily manure storage is a concrete box, with capacities ranging from 155 m^3 to 1,130 m^3, depending on the number of animals served. The facilities are often roofed to eliminate excess capacity otherwise required for precipitation. Another typical facility for poultry and dairy manure storage is an impervious pad (either asphalt or concrete) with earth berm walls.

Both facilities are generally intended for solid or semi-solid animal wastes. These storage facilities also require transfer systems to move manure into the containment area during the winter and to facilitate cleaning in late spring for field application. In addition to storage facilities for animal waste, runoff diversion structures are often necessary in barnyard areas to reduce phosphorus transport. Given the variability of site topography and layout of barns, each farm requires an individually designed manure management system.

In March 1977, the Cobbossee Watershed District estimated the costs of agricultural waste management practices necessary to restore Annabessacook and Cobbossee Lakes and Pleasant Pond. The estimated cost for 41 farms totaled $282,500. Of this amount, 23 major farms would require $266,000 worth of facilities, an average cost of more than $11,500 per farm. Furthermore, construction costs were expected to increase substantially each year because of inflation. Given the precarious economic position of many small farms in Maine, regulatory requirements for agricultural nonpoint source control did not seem feasible.

Financial aid for constructing agricultural pollution controls was limited. The U.S. Agricultural Stabilization and Conservation Service (ASCS) provided a maximum of $2,500 per year (now $3,500) for agricultural conservation practices on each farm. Low interest loans and tax relief measures were also available (Moore, 1979), but generally had not been used for high-cost practices such as manure storage. Thus, the District established a new program of agricultural cost-sharing as part of its lakes restoration effort.

The District offered 50 percent cost-sharing for manure management systems on target farms. The EPA Clean Lakes grant provided the District's cost-sharing funds. The remaining half of construction costs, paid by the participating farmers, provided the 50 percent non-Federal match required by EPA. By using their own labor and materials, participating farmers were able to reduce their actual cash outlays even further. The District's cost-sharing program had no preset ceiling on funds available per farm. Rather, detailed cost estimates for the recommended agricultural waste management plans were developed as the basis for cost-sharing agreements. Thus, individual farms could receive $25,000 in a single year for a $50,000 facility if necessary. Also, certain equipment not usually funded through the traditional ASCS cost-sharing program could be included in the District's program.

Despite the vastly improved cost-sharing ratio, participating farmers were being asked to make significant personal investments in agricultural pollution control. Substantial assistance from the District and various agricultural service agencies was required to achieve voluntary participation. The Cobbossee Watershed District and two county soil and water conservation districts (SWCD's) presented numerous design options to interested farmers. The Kennebec

County SWCD sponsored tours of existing manure management facilities for interested farmers. The Cooperative Extension Service provided data on the economic benefits of conserving manure for use as fertilizer. Personnel of the Watershed District and the SWCD's also assisted with obtaining construction bids from local contractors, information on ASCS, Farmers Home Administration, and Small Business Administration financial assistance, and information on State and Federal tax requirements relating to cost-sharing and pollution control investments. The U.S. Soil Conservation Service (SCS) finalized blueprints and management plans, and inspected projects under construction.

As with any experimental program, unforseen delays developed. The District's cost-sharing program was originally intended to be administered by ASCS, which has many years' experience in agricultural cost-sharing. However, Federal regulations made transferring Clean Lakes funds to ASCS extremely difficult. Thus, the District had to establish its own administrative procedures for cost-sharing. And, instead of developing a single manure management plan for each farm, the District and SCS had to produce three to five detailed alternatives before farmers agreed to participate. These problems, as well as administrative grant requirements, construction scheduling, and other factors, extended the project period from 2 to 3½ years. The extended period has had the significant benefit of allowing a greater number of farmers to participate in the program.

To date, 30 separate agricultural waste management facilites have been constructed. These facilities provide manure storage for approximately 80 percent of the animal units in the watersheds of the three lakes. The cost of these facilities is estimated to be $622,000 (original estimates were revised through grant amendments in 1978). The Cobbossee Watershed District is presently monitoring water quality to determine the reductions in phosphorus resulting from these controls.

NUTRIENT INACTIVATION TREATMENT

Effective restoration of Annabessacook Lake was determined to require reduction of phosphorus recycling from anoxic lake bottom sediments. Various in-lake restoration techniques were considered prior to selecting hypolimnetic treatment with aluminum. The primary concern was controlling phosphorus release from sediments in a 150-hectare area between 7 and 14 meters in depth. Because of the depth and area, dredging, hypolimnetic aeration, and physically sealing the lake bottom were found to be either impractical or prohibitively expensive. Furthermore, flushing of nutrient-rich water from Annabessacook would increase the difficulty of restoring eutrophic Cobbossee Lake, immediately downstream (Gordon et al., 1977).

Prior to implementing a nutrient inactivation treatment, a detailed feasibility study of the project was made (Dominie, 1978). This study attempted to further define the area of bottom anoxia, optimum aluminum application rates, and potential impacts on aquatic life. A detailed discussion of this study and the subsequent treatment is presented elsewhere in this symposium (Dominie, 1980).

Based on the recommended application rates, approximately 227,000 liters of aluminum sulfate and 142,000 liters of sodium aluminate, with a combined weight of approximately 450 metric tons would be needed to treat the lake. Chemical treatment on this scale presented significant logistical problems, particularly with a limited project budget. Because of limited funds, the cost of the treatment was limited to $65,000. As with the agricultural construction, local contributions of in-kind services were used to maximize the Federal funding applied to the project. The Maine Department of Environmental Protection contributed manpower for lake monitoring, development and evaluation of the treatment methods, and laboratory analysis of water samples. Lakeshore property owners affiliated with the Annabessacook Lake Improvement Association and the Cobbossee Yacht Club, provided financial assistance and participated in the lake treatment itself. The Maine National Guard transported a barge for the treatment from Portland to Annabessacook Lake. Boats for the project were donated by the Maine Department of Environmental Protection and lakeshore property owners.

The nutrient inactivation treatment for Annabessacook Lake was completed during August 1978. Despite

Table 4. — Annabessacook Lake visibility.

Secchi disk depth	Public perception of water quality	Days at given visibility (June 1 — September 19) 1972	1977	1979
0 — 0.9 meters	gross pollution; lake is totally unusable for recreation	43	10	0
1 — 1.9 meters	algae blooms still evident; quality is unacceptable for most uses	63	67	0
2 — 2.9 meters	some complaints of declining water quality; some impairment of water use	0	28	30
3 — 3.9 meters	satisfactory quality; no impairment of water use	0	28	30
4 — 4.9 meters	excellent water quality; a positive factor encouraging lake use	0	0	35
5 + meters	exceptional quality for this lake	0	0	1
	Total:	106	106	106

notes on selected years:
1972 — prior to full diversion of municipal/industrial wastewater
1977 — prior to lakes restoration project
1979 — after agricultural waste controls and nutrient inactivation treatment

frequent delays caused by equipment problems, approximately 95 percent of the designated project area received treatment. A total of 179,334 liters of aluminum sulfate and 121,039 liters of sodium aluminate were applied to the lake.

PROGRESS ON LAKE IMPROVEMENT

Monitoring of water quality improvements resulting from the agricultural waste management and nutrient inactivation treatment projects will continue throughout 1981, when a final report will be submitted to EPA. To date, the three lakes affected by the project have not had adequate time to fully respond to nutrient loading reductions it produced. However, preliminary results show progress, with Annabessacook Lake exhibiting the most significant improvement thus far. Phosphorus loading has been reduced by approximately 50 percent, producing a marked improvement in Secchi disk visibility (Table 4). Preliminary data for the summer of 1980 parallel the 1979 results. Reduced phosphorus concentrations in the lake's hypolimnion indicate some effectiveness of nutrient inactivation.

Water quality data for 1979 indicate little improvement in Cobbossee Lake and Pleasant Pond. Tributary sampling indicated continuing phosphorus runoff from farms without proper animal waste management systems (King, 1980). Subsequent construction of manure storage facilities on most of these farms should reduce phosphorus loadings and improve lake quality in 1980-81.

The three lakes are expected to remain sensitive to increased phosphorus loadings even after the full effects of the restoration project are realized. Increasing development in the watersheds of these lakes, perhaps caused in part by their improved water quality, is likely to result in additional stormwater runoff and phosphorus loading. By 1995, the three lakes are projected to once again exceed their phosphorus loading limits, unless additional preventive measures are taken (Gordon, et al. 1980). The Cobbossee Watershed District is planning to concentrate its efforts during the 1980's on preventing any significant deterioration in the quality of its restored lakes, as well as all other major lakes and ponds within its jurisdiction.

CONCLUSIONS

The success of the Cobbossee Watershed District in implementing a lake restoration project can be attributed to integrating technical and financial resources from many sources. Development of lake restoration technology is not enough, if institutional mechanisms to finance and use it are inadequate. The District's efforts through the 1970's illustrate both the opportunities and challenges presented in making the Clean Lakes Program work on the local level.

To control sources of pollution, the District has used non-regulatory approaches whenever possible, attempting to develop a sense of local responsibility for pollution control and lakes management. Public concern about lake water quality led to the creation of the Cobbossee Watershed District. Active citizen involvement in the District's programs has been essential to its success. This public participation will be even more vital in the perpetual struggle to preserve lake quality for future generations.

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THE EUTROPHICATION STORY SINCE MADISON, 1967

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ABSTRACT

The International Symposium on Eutrophication at Madison, Wis. in 1967 summarized current knowledge and provided recommendations for future action. Many of these recommendations have given direction to recent research. This paper examines notable accomplishments during the subsequent 13 years and emphasizes the need to enunciate a new challenge. Accomplishments examined fall in three major categories: (1) Understanding the eutrophication process; (2) developing methods to impede eutrophication; and (3) establishing laws, regulations, and programs to help restore and protect lakes. In the first category, research has focused on: (a) Critical nutrients and the question of carbon significance; (b) nutrient loading and new knowledge derived from the OECD North American Project and the EPA National Eutrophication Survey; and (c) the utility of algal assays in understanding phytoplankton dynamics. To impede eutrophication, methods being tested include: (a) Nutrient manipulations such as diversion, waste treatment, product modification, nutrient inactivation, dilution and flushing, and plant harvesting; (b) physical actions such as aeration, dredging, and hypolimnetic withdrawal; and (c) symptomatic treatments and biological controls. Several laws, regulations, and international agreements have been adopted at various governmental levels to turn back the eutrophication clock. The Great Lakes Agreement of 1972 is one of them. Section 314 of the Federal Water Pollution Control Act provides financial incentives and other mechanisms for lake improvement projects.

INTRODUCTION

Just a few weeks ago, the population of the United States passed the 222 million mark. Today there are 42 million more people than there were when we gathered in Madison in 1967 to share what we knew about eutrophication and to plot new, exciting paths to follow. This past year, 1 billion acres of land were taken up for urban development to meet the needs of population growth. As our Nation moved in these directions, more lakes were caught in the urban fringe as city growth engulfed them; many lakes were newly impacted by sewage effluents in the face of growing demands for recreational use by urbanites.

Are there more eutrophic lakes today than in 1967? I expect so. Are lakes becoming more eutrophic than in the past? Undoubtedly some are, but on the average we don't really know. We do know this — in the lower 48 States there are some 12,000 to 15,000 lakes larger than 40 hectares. They are susceptible. Perhaps 10 to 20 percent are eutrophic, especially ones near urban development that have been sullied by human indifference. Many are well known, and they stimulate and strengthen our concern for the eutrophication problem.

The International Symposium on Eutrophication at Madison, Wis. in 1967 (Natl. Acad. Sci. 1969) assembled the workers and coalesced existing knowledge on eutrophication processes and controls. The document, "Eutrophication — A Review" (Steward and Rohlich, 1967) also appeared in the same year. These developments, and others perhaps less well known, mark 1967 as a most notable reference point for this subject.

In looking at the Eutrophication Story Since Madison, 1967, I am encouraged for several reasons: Public awareness of the problem has grown, many remedial programs unthinkable 13 years ago are underway, anti-eutrophication laws have been passed, and significant new knowledge, ideas and tools, help us probe the eutrophication process. But, I am also disappointed. The Environmental Protection Agency's National Eutrophication Research Program that once was an energetic and moving force, has withered away and no longer exists. We are losing our best opportunity to learn how a lake responds when heroic sewage treatment cuts off phosphorus input. Of course, I'm referring to Minnesota's Shagawa Lake where studies have been terminated before the answers were obtained. Remedial measures available for demonstration in today's restoration programs are the same ones we talked about 13 years ago. I am moved to ask: Where are the new, novel, and exciting ideas that we need now to carry us forward again? Perhaps they will come from what you do here this week.

It is not possible for me to discuss or even cite all important developments since 1967. I have therefore selected examples that are indicative of research trends, control technology currently being used, and regulatory actions that seem characteristic of the past 13 years. Some are the product of research that began much longer ago; others the result of recent beginnings. In checking these examples with the research recommendations that issued from the Madison conference, we can truly say that the conference was a strong inspiration for the years that followed.

There are three main areas of emphasis as I trace this brief history: (1) Understanding the eutrophication process; (2) developing methods to impede eutrophication; and (3) establishing laws, regulations, and programs to help restore and protect lakes.

UNDERSTANDING THE EUTROPHICATION PROCESS

Nutrients

Today, as at Madison, one can still ask: What causes eutrophication? The answer is fragmentary because many interacting factors contribute to the overall process, and how they do so is not always known. Productivity depends on a complex interplay of solar radiation, temperature, lake basin morphology, water retention time, biotic interactions, and perhaps more important, the availability of adequate nutrients. It is generally agreed that algae and higher aquatic plants require many different nutrients for growth, including large amounts of carbon, nitrogen, hydrogen, phosphorus, and smaller amounts of approximately 25 others.

Obviously, rational control depends on somehow interfering with the free action of one or more of these factors. If we seek to starve the system, nitrogen and phosphorus claim special interest because oligotrophic lakes frequently are phosphorus limited; whereas, in lakes enriched by urban sewage, the newly supplied abundant phosphorus often leads to exhaustion of nitrogen. The crucial question is not whether a eutrophic lake is momentarily phosphorus-limited but whether it can be made so through controlling phosphorus input. Over the past 13 years, this has come to be generally recognized.

Then, in the early 1970's, a major controversy arose concerning the relative importance of carbon, nitrogen, and phosphorus. One view contended that carbon is really the regulator of algal production in many waters — a view implying that control of phosphorus input is falsely-based and doomed to failure. The opposing side saw phosphorus as a critical nutrient, the most logical one to be controlled. The heat of the controversy appeared to be fueled by proposals to remove phosphorus from detergents as a step in slowing down cultural eutrophication. In February 1971 a symposium on Nutrients and Eutrophication: The Limiting Nutrient Controversy, was sponsored by the American Society of Limnology and Oceanography. There was free and lively debate in an effort to provide to the public a clear scientific statement on the relative importance of various regulating or limiting nutrients. The symposium ended in apparent general agreement that phosphorus is the critical limiting nutrient in most North American lakes and is the logical focal point for management programs (Likens, et al. 1971; Likens, 1972). Today, the so-called "carbon controversy" seems to have faded away.

Algal Assays

For many years scientists used assay procedures of their own design to estimate the phytoplankton production capacity of lakes and to seek guidance for control procedures. By manipulating their tests in various ways they could also identify critical nutrients representative of the sample at the time. These assays were valuable tools, but unfortunately the findings of one worker could not be compared with another, nor one lake with another, one test with another — and there was little agreement on how to correct this dilemma.

Less than a year following the Madison conference, a small group concerned with this problem met in Chicago under sponsorship of the Joint Industry - Government Task Force on Eutrophication (Anon. 1969). Their purpose was to jointly develop a research plan to produce an algal assay procedure that would be acceptable in North America and Europe and hopefully worldwide (U.S. EPA, 1971). Nine organizations participated in the task — four universities, four industries, and EPA. That goal now seems to have been reached. The bottle procedure, using *Selenastrum capricornutum* as the test alga, has received broad acceptance here and in 41 other countries.

Field Studies

Several large scale field programs have contributed substantially to improved understanding of the eutrophication process. Almost 10 years ago, the Organization for Economic Cooperation and Development (OECD) initiated a study to formulate the relationships between nutrient loadings to lakes and their trophic response. The deliberations were based largely on data available from European lakes. From this effort came the early Vollenweider model concerning nitrogen and phosphorus as factors in eutrophication (Vollenweider, 1968). With time the program broadened in both geography and scope. It began to collect comparable data on the degree and extent to which nutrient loading correlates with eutrophic state, and to measure the rate of eutrophication growth. A major element of the program was the North American Project with specific objectives to: (1) Develop detailed phosphorus and nitrogen budgets for a number of water bodies; (2) assess their chemical, physical, and biological characteristics; (3) relate their trophic states to the nutrient budgets and to limnological and environmental factors; and (4) synthesize an optimal strategy to control the rate of eutrophication. In the U.S. effort, 22 water bodies were studied, and a final report has been published for each (U.S. EPA, 1977). A summary analysis of these reports (Rast and Lee, 1978) gives several points of importance: (1) The Vollenweider nutrient load relationships correlate well with assigned trophic states; (2) good correlation exists between phosphorus loading, normalized as to hydraulic residence time and mean depth, and the average chlorophyll and water clarity; and (3) these relationships can be used to help predict improvement to be expected from controlling phosphorus when that is the critical nutrient.

Simultaneously, a National Eutrophication Survey was underway by EPA to compile information on nutrient sources, inputs, and impacts on selected lakes and reservoirs, especially ones receiving municipal sewage. For over 5 years, the survey sampled and studied 812 water bodies. For each one, the trophic state was estimated, and the sources and magnitudes of nitrogen and phosphorus inputs established as a step toward judging if reduction in phosphorus loading would be a promising remedial approach. Each lake was sampled three times during the growing season at multiple sites and depths. The 15 to 20 analyses done on each sample included nutrient concentrations, algal types and numbers, algal assay data on productivity potential, and limiting nutrient. A separate report is available for each lake, primarily for local officials to use as a starting point for lake restoration program planning. Data summaries are available in four volumes. (U.S. EPA,1978)

Never before has there been such a broad base of similarly-collected data on so many lakes as that now provided by the North American Project and the National Eutrophication Survey (U.S. EPA, 1978). It is noot surprising that these data have been sought and manipulated so avidly in efforts to find the true meaning they may contain. Fear that such interpretations may exceed the validity of the data caused the Ecology Advisory Committee of EPA's Science Advisory Board to take a critical look at the survey and its products. Toward strengthening the credibility of the study the Committee recommended that survey data and evaluation techniques be used to compare well-studied lakes with corresponding ones sampled in the survey. While comparable non-survey data are sparse, especially for tributary point and nonpoint nutrient loads, some limited comparisons could be made. They helped show that the survey data are surprisingly good and certainly adequate to: (1) Assess trophic condition, (2) infer the limiting nutrient, and (3) provide tributary nutrient loads with acceptable accuracy (Allum, et al. 1977).

It has long been recognized that an acceptable nutrient budget is needed for a sound control program. Yet, because of the cost and time required, very few U.S. lakes are even now characterized by such vital information. This will soon change to some extent because lakes can qualify for cost-sharing in the national Clean Lakes Program only if they have a nutrient budget. Major sources of nutrient input such as sewage traditionally received most attention because they were so obvious and easy to quantify. Lesser ones were often ignored or at best only estimated. Today, it is usually recognized that all nutrient inputs are additive and may contribute ultimately to the supply used by plants. Many phosphorus sources now receiving attention include precipitation, droppings of migratory birds, burned gasoline, boats, undisturbed lands, urban land, agricultural land, ground water, industries, and municipal sewage (Bartsch, 1972). There is more emphasis than ever in looking at the total watershed as a significant nutrient source, and the lake and its watershed are increasingly viewed together as intimately connected elements in the management scheme. Maps have been prepared for the United States (Omernik, 1977) that provide a broad overview of nonpoint source stream-nutrient level relationships. While there is no real substitute for a measured nutrient budget, there are now at least ways to provide quick and relatively accurate predictions of nonpoint source stream concentrations of nutrients. With obvious limitations they can serve where more detailed information is not available or resources for specific sampling are not at hand.

The significance of internal phosphorus loading and its impact on lake response to intense recovery efforts has become more sharply appreciated through the studies on Shagawa Lake (Malueg, et al. 1975). This eutrophic lake has been impacted by municipal sewage from the city of Ely, Minn. since 1901. First it was discharged raw, then from 1912 to 1973 with various degrees of treatment. Early in 1973, the input was decreased by about 80 percent through tertiary sewage treatment that yielded an effluent with only 1/20 mg/l of phosphorus. Although the lake improved visually and responded with a prompt and persistent reduction in phosphorus concentration, phosphorus has not declined to the levels expected. The phosphorus residence time model projected an equilibrium concentration of about 12 ug/l within 1.5 years, but it reached only 51. This discrepancy was attributed to feedback from the sediments, primarily during summer.

Modeling

At the Madison conference, the application of modeling techniques to eutrophication was not a prominent discussion subject. In fact, it was hardly mentioned, although research recommendation number 7 urged that "Ecosystem analysis and research on models for simulating trends in eutrophication should be strengthened." Two years later a workshop on Modeling the Eutrophication Process was held at St. Petersburg, Fla. (Anon. 1969), and this was followed by a second at Logan, Utah in 1973 (Anon. 1973). Since then, modeling effort has intensified until a growing array of modifications, adjustments, and substitutions have been made to the nutrient input-output and critical loading models introduced by Vollenweider (1968). Today, models and modeling are routinely used in efforts to better understand the internal workings of specific lakes, to guide regulatory actions, and to anticipate results. Efforts continue on developing more useful models not only for small lakes but for the Great Lakes as well (Thoman, et al. 1979; Ditoro and Matystik, 1978).

METHODS TO IMPEDE OR CONTROL EUTROPHICATION

The Madison conference identified research needed to facilitate control of eutrophication. In particular, research recommendation number 4 urged investigators to seek ways to: (1) Limit nutrient input, (2) accelerate nutrient outgo, (3) impair nutrient availability, (4) reduce the volume of water participating in production of plant material, (5) alter stratification, and (6) modify ecological systems to provide for accelerated consumption of plant material by an appropriate array of animal populations.

Limit Nutrient Input

Because the role of nutrients in eutrophication has been appreciated for a long time, the concept of control through shutting off the supply was a natural step. At Madison, where municipal sewage was discharged to the chain of lakes for many years, the strong voice of the people caused several diversions to take place: First from Lake Mendota in 1899, from Lake Monona in 1936, and from Lake Waubesa in 1958. This is perhaps the longest community struggle with eutrophication in the United States. At Seattle, for the same reasons, effluents from 11 sewage treatment plants were diverted from Lake Washington between 1963 and 1968, and followup studies showed this to be an effective remedial tool. Diversion has been used more recently at Lake Sammamish, Wash., and Twin Lakes, Ohio.

Within the past 13 years, point source nutrient control strategies have shifted to phosphorus rather than whole sewage. Reasons for this are obvious and need no further delineation (Vallentyne, 1970). Two principal phosphorus control strategies have emerged, one concerned with sewage, the other with detergents. Advanced waste treatment to strip phosphorus from sewage was discussed at Madison (Rohlich, 1969), but modification of detergent formulas to reduce phosphorus input to sewage was not on the agenda. The idea of advanced treatment, and the technology for it, are well developed and in many places are keyed to meeting established effluent standards. Modification of detergent formulas to decrease or eliminate their phosphorus content is now required by several jurisdictions. So today, with the focus on phosphorus, several things can be said about standards, advanced waste treatment, and detergents.

Several States, counties, and cities have implemented blanket effluent standards to control point sources of phosphorus. The United States and Canada have jointly adopted standards to protect the Great Lakes. Usually such standards require that treated sewage contain not more than 1 mg/l of total phosphorus. While one can sympathize with the managerial desire for a standard that is technically reachable, financially tolerable, and simple to administer, it ignores the fact that each lake is unique and will respond in its own way. Shagawa Lake helps make this point. Even after an 80 percent reduction of total phosphorus input, improvement has been disappointing. This is because the feedback of phosphorus from the sediments, which has persisted since phosphorus input was first curtailed (Larsen, et al. 1979), is greater than expected. This reaction of Shagawa Lake to an experimental remedial program too costly for practical use, with an effluent standard 20 times more stringent than legally established ones in force, raised a very pertinent question: How appropriate is a 1 mg/l standard? One answer comes from the National Eutrophication Survey. Several of the input-output and trophic state models were applied to data for 225 survey lakes to find how many would benefit under an effluent standard. It was estimated that a 1 mg/l standard would favorably impact only 22 percent of the selected lakes — a zero standard no more than 28 percent (Gakstatter, Bartsch, and Callahan, 1978). This must mean that restoration cannot be accomplished by simply limiting phosphorus input. If phosphorus recycling from bottom sediments is a major factor in the nutrient system, actions to minimize the result would seem to be required.

Added to this concern is the question of how well advanced waste treatment plants remove phosphorus to satisfy the standard. Give or take a little, untreated municipal sewage contains an annual phosphorus load of about 1.4 kilograms per capita. Conventional waste treatment processes reduce this amount by about 36 percent, while phosphorus removal processes can bring it down by about 68 percent or more. Effluents of 809 sewage treatment plants were sampled during the National Eutrophication Survey. Of 33 plants using phosphorus removal processes, the median effluent concentration of total phosphorus was found to be 1.8 mg/l — nearly twice the usual effluent standard (Gakstatter, et al. 1978).

Since the end of World War II, about half the phosphorus in municipal sewage has come from detergents. Decreasing or eliminating this source has been found to be almost as effective as advanced waste treatment in reducing effluent phosphorus. At Onondaga Lake, N.Y. for example, a detergent law limiting phosphorus to 8.7 percent was followed by a 54 percent decrease in inorganic phosphate in treated sewage discharged to the lake. Average concentrations in the lake decreased by 57 percent, and *Aphanizomenon* disappeared during the first growing season.

Accelerate Nutrient Outgo

Harvesting a lake's production to help curb eutrophication through retrieval of nutrients has emphasized macrophytes because effective weed harvesting equipment has been available for many years. Recent attempts to harvest planktonic algae in California have not proved practical. Today, most weed cutting is still a manicuring exercise with beneficial effects sometimes persisting the following year (Kimbel and Carpenter, 1979). Obviously, cutting weeds removes some measure of nutrients because aquatic plants contain some minimal amounts. But, as a method to control eutrophication by limiting nutrients, the real accomplishment is not impressive. Removal of 428,000 kilograms of plants from Sallie Lake, Minn. retrieved less than 1.5 percent of the phosphorus entering the lake (Peterson, Smith, and Malueg, 1974). Recent experiences elsewhere (Burton, King, and Ervin, 1979) have also shown that even the greatest potential harvest will not remove sufficient nitrogen and phosphorus to offset moderate to heavy loading. The outlook might differ if harvesting were an adjunct to cutting off phosphorus input. For whatever reasons, mechanical plant removal is used in only five of 102 U.S. lakes now being restored in the federally funded Clean Lakes Program.

In some ways dredging is an extension of harvesting; one of its goals is to remove nutrient-laden sediments to prevent recycle of their nutrients to the overlying waters. Another frequent goal is to deepen the lake basin to control macrophytes and improve freedom of boat movement. Many U.S. experiences with dredging in recent years seem to have given favorable results but high cost impedes its wider use. Nevertheless, there is a growing interest in dredging as a restoration technique, and more than half the lakes scheduled for restoration in the national Clean Lakes Program will use dredging alone or in conjunction with other procedures.

There are two well known successful examples of using dilution and flushing to cope with the symptoms of eutrophication. At Green Lake at Seattle, Wash., the first introduction of nutrient-poor water from the city's domestic water supply in 1962 (Ogelsby, 1969) produced striking improvement. The program has continued since that beginning. Success here led to a similar test program at Moses Lake, Wash. where dilution water from the Columbia River was introduced in spring and summer of 1977 and 1978 (Welch, 1979) and several times since. Impressive reductions in total phosphorus and chlorophyll *a* concentrations resulted and Secchi disk depths increased strikingly. Obviously, this simple approach to curbing eutrophication is exceedingly attractive and is being further tested in four other U.S. lakes. The practical barriers are lack of large supply of high quality water, absence of physical structures to introduce it, and need for sympathetic people residing downstream who are not affronted by the prospect of receiving the "flushings."

At Snake Lake, Wis. nutrient-rich water was once pumped out to permit the seepage inflow of higher quality ground water. This novel approach has not attracted much attention and, to my knowledge, has not been used elsewhere.

Where the physical setting permits, hypolimnetic withdrawal can be used to accelerate nutrient outgo and improve dissolved oxygen conditions near the bottom. In reservoirs, where selective depth withdrawal controls may be available, deep withdrawal may be a choice approach. In lakes, equipped with only surface exits, nutrient-rich water must be removed by pumping or siphoning from the point of maximum depth. This is not a well-known technique but has been used in several States.

Impairing Nutrient Availability

Two approaches that reduce the availability of nutrients have been used with some success. The first involves chemical treatment of lake water *in situ* to precipitate phosphorus — a nutrient inactivation approach apparently first used at Langsjon, Sweden in 1968. Aluminum sulfate or other aluminum compounds has since been used in many bodies of water ranging upward in size from Cline's Pond, Ore. at 0.4 hectares to Liberty Lake, Wash. at 277 hectares (Funk and Gibbons, 1979). With few exceptions the treatments have reduced phosphorus concentration, limited nuisance algae, and helped maintain adequate oxygen. At least nine U.S. lakes are being treated by chemical nutrient removal. Research since 1967 has emphasized improving procedures and equipment and searching for more effective inactivating agents, including small field tests with zirconium and lanthanum compounds.

The second approach seeks to immobilize nutrients through aeration of hypolimnetic water where large reservoirs of phosphorus reside. Equipment and procedures have been developed to permit aerating only the hypolimnetic water without destratifying the lake. This can be accomplished by injecting air or pure oxygen or by mechanical means (Fast, 1979). As a result, nutrient upwelling is minimized and suitable temperature preserved for cold water fisheries. When the method is designed to destratify, the lake becomes isothermal with oxygen available to the bottom, and other chemical conditions are fairly uniform. Both types of aeration have been used in Europe and North America but are not currently popular in the national Clean Lakes Program.

Reducing the Volume of Water Participating in Production of Plant Material

During the past 13 years, learning to reduce the volume of water that participates in plant production has been largely ignored. New ideas have not emerged. One or two historical trials come to mind. In one, decreasing the volume of the photic zone was attempted by treating two Arizona ponds with the dye nigrosine (Eicher, 1947). The reduction in light penetration impaired growth of semi-emergents for a few years. In 1977 analine dyes were used successfully in Nebraska farm ponds (Buglewicz and Hergenrader, 1977). In another trial, weed-choked Deer Lake, N.J., was treated with commercial fertilizer to stimulate increased production of phytoplankton (Surber, 1948). When sufficiently dense they served as a sun shield and proved successful in curtailing plant growth. With that purpose accomplished, the lake was drawn down to dispose of the enriched water. Even with this success, it is doubtful either one could stand the rigors of today's environmental impact scrutiny.

Accelerate Consumption of Plant Material

Manipulating biological interactions to benefit lakes is best known in fishery management. Its use in alleviating symptoms of eutrophication was mentioned at Madison and expanded research suggested. But biomanipulation can only mature as our basic knowledge of biota and biological interactions becomes more complete. For now we have few triumphs to exhibit. We can only point to a few herbivores with voracious appetites that drive them to attack specific plants; for example, the flea beetle that devours alligator weed, another insect that eats water hyacinth, and the grass carp. Crayfish, snails, swans, and manatee that were once viewed as promising candidates, do very little.

Not much has been done to control algal populations through biological means. Microorganisms that destroy blue-green algae were isolated a long time ago. Although promising in laboratory tests, no full scale lake treatments have been tried in the United States. Interest in the so-called blue-green algal viruses appears to be swinging upwards again. Recently, Shapiro (1979), in championing biomanipulation, urged us to not be so hypnotized by the easy use of phosphorus loading models that lake biology is ignored. Certainly, the admonition is worth your consideration.

LAWS, REGULATIONS, AND PROGRAMS

It is safe to say that in the United States the past 13 years have produced more eutrophication superlatives than all preceding history: (1) More dollars spent to study the subject, (2) more dollars devoted to more lakes for restoration, (3) more laws and regulations to expedite correction.

Adoption and current upgrading of a phosphorus-control plan established under the Great Lakes Water Quality Agreement between the United States and Canada is one of the two most important and far-reaching milestones since 1967. The other is the passage of the U.S. Clean Lakes Program legislation in 1972 and startup of the program in 1975.

Ten years ago the International Joint Commission alerted the governments of Canada and the United States to the accelerating eutrophication in Lakes Erie and Ontario and cited the danger of permitting unabated nutrient inputs. The two countries responded by signing the Great Lakes Quality Agreement on April 15, 1972 to jointly implement programs to reduce phosphorus loads entering the Great Lakes System. These programs, which have made an impressive start in these 8 short years, focus mostly on point sources such as municipal sewage, industries, animal husbandry operations, and detergents. Recognizing the need to focus on diffuse sources as well and to attain more stringent phosphorus load targets led to a new agreement in 1978. A recent draft report (Phosphorus Manage. Strat. Task Force, 1980) now outlines a proposed updated plan for phosphorus management in the Great Lakes. The plan is currently under consideration by both countries.

Amendments to the Clean Water Act (P.L. 92-500) passed in 1972 set the stage for a massive national effort to protect and restore lakes. The resulting Clean Lakes Program seeks to remedy in-lake problems and control nonpoint source pollution in the tributary watersheds. Local interest has been intense, largely because matching funds are available to help cover the cost of lake restoration projects. The program thus sets the stage to demonstrate and evaluate a wide array of remedial technologies. Unfortunately, the technologies currently contemplated do not reflect a new giant step forward since 1967. They include such familiar approaches as hypolimnetic destratification, bottom sealing, biofiltration, biomanipulation, chemical nutrient removal, and flushing. Projects are now underway or imminent in 102 impacted lakes located in 28 States at a total cost of about $90 million. Each project must be given a followup evaluation to record success or failure, but 12 lakes are receiving in-depth study over a period of years to sense lake response, durability of improvement, and to gain a better understanding of why the lakes responded as they did.

As part of this effort, States are required by law to classify all publicly owned lakes as to their trophic condition and to identify causal factors. Only a few States have completed the task but this has stimulated study of earlier classification schemes and development of new alternates. Unfortunately, there is not yet a uniform scheme for trophic classification.

These actions to protect precious lake resources were built upon a legislative beginning which, like so many lacustrine developments, began at Madison many years ago. One of this country's earliest pieces of legislation to address the nutrient input problem is Wisconsin's Lewis Bill. Enacted just before the end of World War II, it was carefully designed to prevent Madison's effluent from reaching the chain of Madison lakes. It was never enforced because it was judged legally void in 1949. Nevertheless, its purpose was ultimately attained, and the lakes are now protected. Jurisdictions throughout the country have since passed laws constraining nutrient loadings. Most either specify maximum allowable amounts of phosphorus in treated sewage that reaches susceptible waters, or they set maximum amounts of phosphorus for detergents.

In Minnesota, if a discharge from a sewage treatment plant enters a lake directly, the phosphorus content must not exceed 1.0 mg/l; if it reaches a lake via a river, up to 2.0 mg/l are allowed. Illinois has an effluent standard of 1.0 mg/l for discharges that flow to Lake Michigan. Other States have similar standards.

Laws regulating detergent phosphorus were passed in New York and Indiana in 1971. Both required total elimination of phosphorus from detergents by specified dates in 1973. Laws of the same intent were also passed in Florida, Maine, Michigan, Minnesota, Connecticut, and Oregon, as well as Chicago, Akron, and Dade County, Fla.

A 1971 Iowa law required mandatory soil conservation, viewing as a nuisance soil erosion that causes siltation damage. That erosion damage is not in the lost soil alone but often in the silted lake as well is now more appreciated. Funding of National Soil Conservation Programs is guided by this fact.

CONCLUSION

In conclusion, I wish to leave three points with you:

First, as I look at the lake protection and restoration technology in use today, I am convinced there is room for substantial improvement. I hope the discussions you enjoy here this week will identify many new ideas that can be pursued.

Second, scientific curiosity will continue to provide better answers to the question: "What causes eutrophication?" As the new answers emerge, so will the prospects to develop the improved technology needed for more effective lake management in the years ahead.

Third, human attitudes, often of people in powerful places, must be changed if lakes generally are to be protected or restored. Four years ago, several colleagues and I used the following words to introduce a paper on the status of eutrophication in the United States (Bartsch, et al. 1978). They were spoken by Chief Seattle of the Suquamish tribe in Washington Territory in 1854 when agreeing to transfer Indian land to Federal ownership.

"This shining water that moves in the streams and rivers is not just water but the blood of our ancestors. If we sell you land, you must remember that it is sacred, and you must teach your children that it is sacred, and that each ghostly reflection in the clear water of the lakes tells of events and memories in the life of my people. The water's murmur is the voice of my father's father.

"The rivers are our brothers, they quench our thirst. The rivers carry our canoes, and feed our children. If we sell you our land you must remember, and teach your children, that the rivers are our brothers, and yours, and you must henceforth give the rivers the kindness you would give any brother.

". . .The earth does not belong to man; man belongs to the earth."

In light of the historical record, these pleading words are even more timely today than when they were first spoken 126 years ago.

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NORTH AMERICAN OECD EUTROPHICATION PROJECT: THE UNITED STATES STUDY

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ABSTRACT

The U.S. portion of the North American Project included 34 water bodies ranging from ultra oligotrophic to hypereutrophic. The U.S. OECD study consisted of gathering, analyzing, and synthesizing existing eutrophication-related water quality data from water bodies which had been intensively studied, rather than conducting new field studies. It was determined that the Vollenweider nutrient loading relationship correlated well with the trophic states identified by the investigators. After the study, approximately 40 additional water bodies were found to exhibit the same basic phosphorus load-response relationships. A summary of the original U.S. OECD study as well as subsequent studies is presented.

The United States portion of the North American Project consisted of 34 water bodies located primarily in the north central and northeastern United States. In contrast to the other Projects, the U.S. OECD water bodies were not of one specific type, but rather exhibited a range of trophic character from ultra-oligotrophic to hypereutrophic. Further, the United States participation consisted of gathering, analyzing and synthesizing existing water quality and other data related to eutrophication from water bodies which had already been extensively studied, rather than conducting new field studies as was done in the other OECD Projects. The U.S. OECD Study was completed before the other Projects.

A summary analysis on the U.S. portion of the North American Project was prepared by Rast and Lee (Summary Analysis of the North American (U.S. Portion) OECD Eutrophication Project: Nutrient Loading — Lake Response Relationships and Trophic State Indices, Ecological Research Series, EPA-600/3-78-008, 1978). The individual lake studies were reported in a standardized format and were compiled by Seyb and Randolph (North American Project: A Study of U.S. Water Bodies, Ecological Research Series, EPA-600/3-77-086, 1977).

The U.S. Environmental Protection Agency was the lead agency for the study in the United States. Emphasis was on the development and use of quantitative lake management models for assessing eutrophication and the effects of phosphorus control programs, using the statistical nutrient loading models of Vollenweider as an initial focus.

The 34 water bodies in the U.S. study included 24 lakes, nine impoundments and one estuary. When sub-basins of these water bodies were considered, there were 37 distinct water bodies in the U.S. study. The principal investigators for the individual water bodies classified 25 as eutrophic, five as mesotrophic, and seven as oligotrophic as of the completion of the study. Twenty-eight water bodies had mean depths less than 10 meters (range = 1.7 to 313 m), while 16 water bodies had surface areas greater than 1,000 hectares (range = 47 to 1.7×10^7 ha). Twenty water bodies had hydraulic residence times greater than 1 year (range = 0.08 to 700 yr), while 28 had Secchi depths less than 3 meters (range = 0.6 to 28 meters). The general morphometric, hydrologic, chemical, and biological characteristics of the U.S. OECD water bodies, as well as other pertinent data, are summarized in the original Rast and Lee report.

Components of the summary analysis of the U.S. study included an examination of analytical procedures for major biological or chemical water quality parameters, determination of the limiting nutrient, and evaluation of methods for the identification of major nutrient sources and for the calculation of nutrient loads. The nutrient load estimates provided by the U.S. OECD investigators were compared with estimates derived on the basis of the Vollenweider model relating influent and in-lake phosphorus concentrations, and with estimates based on nutrient export coefficients and land use patterns within the U.S. OECD water body watersheds.

In general, it was found that the phosphorus and nitrogen load estimates for the water bodies were within a factor of ±2 of the load predicted on the basis of the Vollenweider approach and the nutrient export coefficients. Possible reasons for any anomalous load estimates that were encountered were investigated. Phosphorus residence times were also investigated and were generally found to be shorter than the hydraulic residence times, usually by several-fold, being shortest as the degree of eutrophy increased.

It was determined that the results of the Vollenweider nutrient loading diagram relating between annual areal phosphorus load and hydraulic load correlated well with the trophic states identified by the individual investigators for the U.S. OECD water bodies. The similar positions of the water bodies on a nitrogen loading diagram as on the phosphorus loading diagram indicated a relative constant ratio of nitrogen to phosphorus loading to the water bodies.

Using the phosphorus load - chlorophyll model of Vollenweider as a guide, a statistical correlation was developed between phosphorus loading, normalized by mean depth and residence time, and chlorophyll *a* concentrations in the U.S. OECD water bodies. The chlorophyll - Secchi depth relationship in water bodies was also examined and used to derive a direct relationship between normalized phosphorus load and Secchi depth. A statistical correlation was also developed which directly related normalized phosphorus load and hypolimnetic oxygen depletion rate. These models are presented in graphic form in the summary analysis of the U.S. study.

Several trophic status indices were also compared using the U.S. OECD water bodies as a data base, and were found to predict relatively identical results. A trophic status index was also developed using the Vollenweider diagram relating annual areal phosphorus load and hydraulic load, thereby relating trophic status to critical phosphorus loading levels. A large number of correlations between nutrient loads and/or various in-lake chemical, biological, and physical parameters in the U.S. OECD water bodies were also examined. The use of different analytical and sampling methodologies and the varying number of data sets for a given correlation, however, limit the general usefulness of these correlations based solely on data from the U.S. study.

Overall, the statistical models developed in the U.S. study can be used to predict the changes in water quality related to eutrophication that will result from changes in phosphorus loads to water bodies for which phosphorus is the key element limiting planktonic algal growth. These models relate the normalized phosphorus load of phosphorus-limited water bodies to several commonly used water quality parameters. The U.S. study indicated the validity of the basic Vollenweider approach for determining the critical phosphorus loading level and associated overall degree of fertility of water bodies. The models developed during the U.S. OECD study offer simple, practical, and quantitative methodologies for assessing the expected effects on water quality of eutrophication control programs based on (1) phosphorus removal from domestic wastewaters, and (2) other phosphorus controls.

Following the completion of the U.S. study, approximately 40 additional load-response relationships in water bodies were evaluated and found to exhibit the same basic phosphorus load-response relationships. The basic approach has also been extended in the development of a correlation between normalized phosphorus loads and overall fish yield. Further, the predictive capability of this statistical modeling approach has been demonstrated by comparing measure changes in water quality response parameters which occurred after phosphorus load reductions to about 18 water bodies, with the changes predicted by the models developed in the U.S. study. A detailed manual on the practical use of the U.S. OECD models has also been prepared. A summary of these subsequent related studies is available from the authors.

MONITORING OF INLAND WATERS: THE NORDIC PROJECT

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INTRODUCTION

Combating eutrophicaton requires a combination of knowledge and money. The eutrophication problem has been of central interest for OECD, the Organization for Economic Cooperation and Development. In 1966 OECD supported a study of existing literature on eutrophication with special reference to the roles played by nitrogen and phosphorus in the process. This report emphasized that relevant measurement data were insufficient to permit more precise guidelines and advice for the control of eutrophication.

In 1973 the OECD Water Management Group initiated a 4-year cooperative program to monitor inland waters. The program was subdivided into three regional projects: the Alpine, Nordic, and North American, plus a non-regional reservoir project. This paper presents results from the Nordic Project in a condensed form. The full report including recommendations to improve and optimize lake management programs and the outcome of using predictive lake models has been published by the project coordinator, the Nordic Cooperative Organization for Applied Research — NORDFORSK (Ryding, 1980).

BACKGROUND DATA

The participation from the Nordic countries consisted of research data from 10 lakes. The lakes differed a lot regarding climate, morphometry, hydrology, and loading conditions. The following ranges for some important background data may be noted:

Height above sea level (m)	0.3	-	103
Catchment area (km^2)	84	-	26,480
Ice coverage (days)	60	-	150
Lake surface (km^2)	2.7	-	1,912
Volume (km^3)	0.02	-	74
Average depth (m)	3.1	-	153
Outflow (m^3·s^{-1})	0.8	-	320
Hydraulic residence time (year)	0.2	-	57
Nitrogen supply (g·m^{-2}Yr^{-1})	1.8	-	101
Phosphorus supply (g·m^{-2}Yr^{-1})	0.1	-	3,6

As a consequence of different land-use patterns of the drainage basins and the different morphometrical and hydrological conditions of the water bodies, water quality varied greatly. A high transparency was found in lakes low in P and algae (chlorophyll) and vice versa. N- and P-concentrations often maintained the same relation to each other whether total concentrations or soluble inorganic fractions (NH_4 + NO_2 NO_3) - N and PO_4 -P were considered. Primary production and chlorophyll were closely related. The hypolimnetic oxygen depletion rates in the Nordic lakes, however, did not seem to correlate to primary production, but the lack of data regarding these parameters in some lakes makes a straight comparison difficult.

The annual nitrogen load was found to be less correlated to the nitrogen concentration in the lake waters compared to the corresponding relationships for phosphorus. The supply of P and its concentration in the lake waters were more strongly correlated to the trophic state of the water body than N, indicating that P can be regarded as a key chemical element limiting planktonic algal growth. P concentrations in lake water were closely related to chlorophyll, based on different annual and seasonal calculations. As a measure of biological response predicted from the nutrient load the parameter chlorophyll *a* was found to be superior to primary production.

The very strong correlation between annual maximum and summer average or annual average concentrations of chlorophyll reveals that a certain basic level of chlorophyll is a prerequisite for peak values to occur.

Adoption of the Nordic data to lake models based on the phosphorus load versus mean depth relationship was somewhat misleading, particularly for lakes with a high flushing rate. Later modifications also taking into account the hydraulic residence time and phosphorus retention predicted the trophic states about equally well for the majority of the Nordic lakes if compared with that subjectively chosen by each project leader. Phosphorus loading diagrams transferred to nitrogen overestimated the trophic states of the Nordic lakes as either a result of a too low conversion factor or the unsuitability of applying data from P-limited lakes into nitrogen-load models.

Predictive models based on nutrient load-lake response relationships are valuable tools for assessing the expected effects of phosphorus reduction, e.g., from domestic wastewater by advanced wastewater treatment or sewage diversion as illustrated for three of the Nordic lakes, on eutrophication control programs, and as a base for establishing phosphorus load and water quality criteria.

SPECIAL HIGHLIGHTS

The treatment of the Nordic data was performed in two ways — "handmade," sometimes excluding outliers before calculations of correlations and load-response relationship, and purely computerized on the complete data set. The approach of treating the whole data set totally computerized did not reveal the overall relationships verified from the other sub-projects of the OECD eutrophication program including the handmade treatment of the Nordic data presented here. In a research program carried out in a diverse group of lakes, it is therefore necessary that data treatment and assessment of the results are made using "biological know-how." Treating biological research data using only a statistical-technical approach may be hazardous.

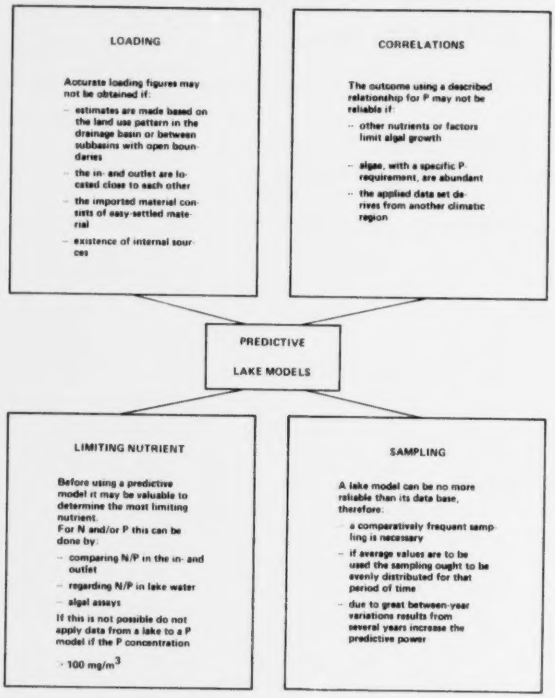
Using algal assay, the algal growth potential, the "free capital of nutrients," was generally found to increase with a higher trophic state. Phosphorus was generally the most limiting nutrient if the total nitrogen to total phosphorus ratio exceeded 13. In waters with lower values nitrogen played a major part regulating algal growth. The corresponding figure if the ratio is calculated for the dissolved inorganic fractions was 12. Trace elements, iron and/or a chelating agent (EDTA) were found to stimulate algal growth in some of the Nordic lakes. Using information on the growth-limiting role of nitrogen and phosphorus obtained by performing algal assays a stronger correlation for the nutrient load-lake response relationships was obtained by adding the growth effects of these nutrients together and an expression for the "load of algal growth-limiting nutrients."

The results from the Nordic project permit a composite model, predicting the summer average and annual maximum concentrations of chlorophyll in a phosphorus limited lake derived from simple empirical findings on phosphorus load and phosphorus concentration in lake water. It is important that the data collected in the OECD study on monitoring of inland waters are used also for evaluation and assessment of the validity of the existing models in lake management. The contribution from the Nordic project to improve the predictive power of lake models is a list of different aspects regarding sampling procedure, loading calculations, the limiting nutrient concept and phosphorus-chlorophyll relationships that ought to be fulfilled to optimize the outcome of using the models. These aspects, graphically illustrated in Figure 1, should be considered as a first-cut analysis to be done before interpreting the outcome from a comprehensive data set being used in the models.

DISCUSSION

It is difficult to define the complex interactions that occur in a body of water to the point where detailed accurate assessments can be made of the impact of a point or nonpoint wastewater source discharge on a lake.

Over the years, two basic approaches have evolved for use by agencies in making decisions on the limitations of nutrients into aquatic systems, i.e., ecosystem models and P-load models. Useful as they might be, for practical and economical reasons complex ecosystem models may not be able to replace the simple parameters as chlorophyll and phosphorus. Furthermore, using even a complex model based on various interactions among the components of the aquatic ecosystem it may be found that some lakes do not fit the model because of special or local conditions. Improved modeling techniques for large-scale lake management schemes must be developed in conjunction with sound methods for making routine measurements of sensitive environmental variables.



Accurate loading figures may not be obtained if:

- estimates are made based on the land use pattern in the drainage basin or between subbasins with open boundaries
- the in- and outlet are located close to each other
- the imported material consists of easy-settled material
- existence of internal sources

The outcome using a described relationship for P may not be reliable if:

- other nutrients or factors limit algal growth
- algae, with a specific P-requirement, are abundant
- the applied data set derives from another climatic region

Before using a predictive model it may be valuable to determine the most limiting nutrient.
For N and/or P this can be done by:

- comparing N/P in the in- and outlet
- regarding N/P in lake water
- algal assays

If this is not possible do not apply data from a lake to a P model if the P concentration

> 100 mg/m^3

A lake model can be no more reliable than its data base, therefore:

- a comparatively frequent sampling is necessary
- if average values are to be used the sampling ought to be evenly distributed for that period of time
- due to great between-year variations results from several years increase the predictive power

Figure 1. — Different aspects that ought to be fulfilled for optimizing the outcome when using predictive lake models.

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OECD EUTROPHICATION PROGRAM REGIONAL PROJECT: ALPINE LAKES

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ABSTRACT

In a coordinated international program, the correlation between nutrient loading of lakes and their biological-chemical (trophic) response was examined in four partial projects. The Alpine project which is described here, dealt with five countries: Austria, the Federal Republic of Germany, France, Italy, and Switzerland. A critical analysis is made of the quantitative assessment of nutrient load. Trophic classification was defined by means of a probability approach. The overturn value of phosphate phosphorus and the maximum chlorophyll concentration were the most significant trophic level indicators. The annual and the maximum daily primary production could be associated with the spring overturn value of phosphorus by a hyperbolic estimate. Apart from simple correlation techniques, empirical phosphorus loading models (elementary mass balance concepts; mixed reactor theory) and modifications of the steady-state conditions for a conservative compound with an additive time-variable term were used. No striking differences were observed among the correlations. On the basis of the correlation found, a lake's reaction to a change in phosphorus load can be predicted to a certain degree. The limits of the applied concept are discussed.

The Alpine project that is described here, is part of the OECD International Investigation Program on Lake Eutrophication. The total program covered approximately 200 natural and artificial lakes, spread around the world. They were grouped into projects according to technical and geographical criteria:

- Alpine project
- Shallow lakes and reservoir project
- Scandinavian project
- US/Canada project

The results of each project are published in a comprehensive report.

The Alpine project dealt with five countries: Austria, the Federal Republic of Germany, France, Italy, and Switzerland. Data on 28 Alpine lakes or lake basins were obtained by voluntary cooperation. Most of the data were calculated or adapted to the purpose of the study by using a unit process. Several ringtests were made to establish parallels between the results of each laboratory, thus refining the method (detailed results are given in the report).

The lakes of this entire region are strongly influenced by their mountainous surrounding, topographically and climatically. Basic criteria for classifying a lake as Alpine are as follows:

- Complex mineralogy: limestones, dolomite, granite etc.
- V-shaped or with rocky, steep side slopes, except those lakes lying in the Swiss midlands and similar flat valleys in Germany (Bavaria) and Italy (Brianza lakes).
- Relatively deep (100 and more meters), and because of this, a special stratification behavior (if the wind exposition of the valley is good (Urnersee), then these lakes can mix fully. Consequently, they have high tolerance level for phosphorus. But in other cases the mountains prevent full circulation (Kreuztrichter, Lago di Lugano) and these lakes tend to become anaerobic in the hypolimnion.) A significant phosphorus input from the sediments is a further consequence.

The final selection of the lakes for the OECD program was influenced by the following points:

- Various trophic states, hydrological residence time, and mixing regime.
- A monitoring program already in operation.

SUMMARY OF THE RESULTS

A critical analysis was made of the quantitative evaluation of the nutrient load. Although the limnological behavior of many lakes with respect to their biological-chemical reaction has become well known, loading measurements have often been neglected. But in the last 5 to 10 years, increasing attention has been given to this problem. A main step forward was made in the OECD Eutrophication program. A deeper evaluation of the assessment of nutrient loading in the Alpine part makes it clear that the influxes had not been sufficiently investigated. Especially water flow and highwater surveillance have been insufficiently measured. (In a special chapter of these Proceedings guidelines are given to encourage limnologists to measure nutrient loading directly to acquire accurate data base).

An important question during the OECD study was the classification of trophic state. An attempt was made to define a trophic state as a system of probability approaches. The overturn value of phosphate phosphorus and the maximum chlorophyll concentration were most significant trophic level indicators.

The main concept of the study was to describe the average behavior of lakes in response to available data on nutrient loads based on simple statistical approaches. The ratio between nitrogen and phosphorus confirms that in most cases phosphorus is in fact the limiting factor. To a certain extent it is possible to calculate the phosphorus concentration in the lakes by the annual inflowing concentration or by the more complex Vollenweider model. A similar correlation can be achieved by the somewhat different Schindler approach.

Primary productivity measurements were unfortunately not given adequate attention, even though this is the only parameter which directly influences the trophic level. Nevertheless, to some extent it is possible to mathematically describe the relationship between primary production and load by using statistical approaches. The annual and the maximum daily primary production could be associated with the spring overturn value of phosphorus in a hyperbolic estimate.

The nutrient loading characteristics determined in the program make it possible, by using simple correlation techniques, to generalize to some extent on the average statistical behavior of lakes in response to their nutrient loading. Statistically, it is possible to predict, with a certain degree of reliability, the average lake concentration of phosphorus and chlorophyll, as well as the average annual primary production.

In general, the connections between load and lake parameters are plausible, although we have to take notice of the fact that they are based on statistics, with a certain deviation and probability. Therefore they may not be applied uncritically for practical purposes and predictions. They need an interpretation in which the peculiarities of the single lake in question must be considered.

Due to technical reasons it was possible to measure only the external input of phosphorus, while the internal load, which is also an essential parameter in lake eutrophication but extremely difficult to determine, was not taken into consideration. For this reason, direct application of the results (e.g. for therapeutical purposes) is not always possible.

It is an established fact that all the technical measures for lake recovery and conservation aim at reducing the phosphorus load. The lake's tolerance load can be calculated by means of this study. In numerous cases, it will be possible from now on to reach this limit of tolerance by external measures (wastewater treatment, ring trunk sewers); in others, however, this will not be possible. Additional protection measures will then have to be taken to decrease or interrupt the internal phosphorus supply (aeration, destratification, discharge of hypolimnetic waters). To quantify these measures, a sound knowledge of the complexity of the internal nutrient cycles is essential, knowledge which the present study is not able to supply. The complex strategy resulting needs planning which is only possible on the basis of a time dependent (dynamic) model. The statistical models of this report are a suitable tool for decisions mentioned here and for political decisions on whether a dynamic modeling is necessary (which is rather time consuming and costly). In any case the statistical model's application is the indispensable basis or the first step to the following decisions.

Some non-scientific but nevertheless most important facts should be considered:

- The friendly cooperation between the laboratories has been an extremely fruitful experience, has created personal friendship and solid mutual confidence.
- By common work, in particular by the calibration tests, the quality of the data has increased significantly.
- The necessities of the OECD program produced enough pressure to set new analytical developments.
- The OECD program has been important enough to set up new research laboratories which today are of great value in the water protection networks of the respective countries.

Abstracted from:

Fricker, Hj. 1980. OECD Eutrophication Program: Regional Project Alpine Lakes. Swiss Fed. Board Environ. Prot. (Bundesamt fur Umweltschutz). CH-3003 Bern, Switzerland.

THE SHALLOW LAKES AND RESERVOIRS PROJECT

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The OECD regional project "Shallow Lakes and Reservoirs" included a large number of extremely varying waterbodies which, for the purpose of the project, were divided into two main groups: natural basins and artificial basins. The natural basins were mainly shallow waterbodies (located in Ireland, Japan, Netherlands and the United Kingdom). The group of artificial basins included pump storage reservoirs, which have been created by construction of ring-shaped barriers (located in Netherlands and United Kingdom), and semi-artificial reservoirs created by impounding natural valleys (located in Australia, Germany, Netherlands, Spain and United Kingdom). Several reservoirs in the United States portion of the North American Project were also included in this project.

Initial data analysis consisted of a statistical survey of the collected data. This survey showed that the values were not normally distributed. Logarithmic scales were suitable for most of the correlation graphs. As expected from the title of this project, the average of all the mean depths was low (approximately 9 meters). If one considers shallow lakes to be defined as lakes in which stratification never occurs, or in which it occurs for only very short periods, then 47 percent of the lakes in this project are shallow. The average retention time of all the lakes under study was approximately 6 months, which is remarkably short. The majority of the lakes (approximately 70 percent) were eutrophic.

If a model is to be developed which describes the relation between algal biomass and nutrient input, it is of fundamental importance to determine whether nitrogen and phosphorus is the limiting nutrient. If the N:P ratio is calculated for every lake or reservoir in the project and compared with the N:P ratio considered to be ideal for algal growth (N:P ≃ 15:1), then the limiting nutrient can be established. The results obtained show that almost no lake or reservoir in this project can be considered nitrogen-limited. Thus, it was possible to apply models decribing the relation between phosphorus supply and trophic state to these waterbodies.

For this purpose Vollenweider's well-known formula for phosphorus loading was generalized, and the coefficients were recalcualted by iteration. This lead to a slightly different relationship, which showed that phosphorus retention in the lakes and reservoirs examined was greater than that calculated by Vollenweider's original formula. This deviation seems to be independent of the lake type in this project since natural lakes and pumped storage reservoirs did not suit the original formula better than semi-artificial reservoirs, to which the "chain of reactors" theory could be applied. This theory assumes that the long and narrow semi-artificial reservoirs can be regarded as a cascade of reactors in which phosphorus is more effectively retained than in one large reactor.

In further analysis of this deviation, it seemed best to first examine the extent to which the phosphorus retention depends on the inflow concentration, disregarding the water residence time. This is done simply by plotting average in-lake phosphorus concentration against average inflow concentration, and correlating the two parameters. A simple power curve was used for regression.

Although water residence time was not taken into account, the correlation was remarkably good. It is significant that in the equation obtained, the coefficient was clearly less than 1. This means that, independent of retention time, a high inflow concentration is generally reduced in a lake to a greater degree than a low inflow concentration. Thus, phosphorus retention is of more importance in eutrophic lakes than in those which are oligotrophic. This is contrary to the opinion that it is in eutrophic lakes most of the phosphorus which reaches the bottom as a result of sedimentation is released again. One should, however, consider the fact that phosphorus is probably more effectly used in oligotrophic lakes, i.e., the algae in these lakes contain less phosphorus than those in eutrophic lakes. In eutrophic lakes, however, algae take up phosphorus in excess of their requirements (luxury uptake), which has long been known to limnologists.

A parameter describing phytoplankton density is chlorophyll *a*, the primary assimilation pigment of all algae. Not only is it easy to determine the chlorophyll content of algae in a relatively simple way, but the technique is also exceptionally sensitive, which means that even low plankton densities in oligotrophic lakes can be determined. In some reservoirs, the total phosphorus-chlorophyll relationship was much lower than expected. This was the case in the reservoirs, "Honderd en Dertig" and "Petrusplaat", and in the Australian reservoir, 'Mount Bold." In these reservoirs, it was not phosphorus, but rather light which was limiting primary production. Therefore, these reservoirs were excluded from calculations of the relationship between nutrient supply and algal.

For determining the correlationship between the chlorophyll concentration and the total phosophorus concentration in the euphotic zone, both Mitscherlich's saturation function and a simple power curve were applied. Very similar results are obtained with both methods. The fact that the power coefficient was less than 1 indicates that the ratio betwen chlorophyll and phosophorus decreases with increasing phosphorus concentrations. It was not possible to show that chlorophyll concentrations depended on total nitrogen concentrations, which was to be expected.

In general, the investigator's evaluations of trophic state was quite consistent with the previously valid border lines in the phosphorus loading models, as well with the recent statistical data for phosphorus and chlorophyll. On the basis of phosphorus only, the Australian reservoir, Mount Bold, deviated considerably from the general border lines defining predicted trophic state. It had been classified as "mesotrophic" by the investigator, whereas on the basis of its phosphorus content, it should be considered "eutrophic" or even "hypereutrophic."

It is interesting to note that Mount Bold can be classified as being mesotrophic or oligotrophic with almost the same probability as if using chlorophyll concentration as the criterion. This is probably because a considerable amount of the phosphorus is not fixed in planktonic algae but instead is in the silt. The Queen Elizabeth-II Reservoir in the United Kingdom did not fit the picture at all. This is because since the total depth was high compared to the euphotic depth, it was possible to keep algal density low by means of artificial circulation.

BACKGROUND AND SUMMARY RESULTS OF THE OECD COOPERATIVE PROGRAM ON EUTROPHICATION

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THE PROBLEM OF EUTROPHICATION

Early in the 1960 decade, it became obvious that a large number of lakes and reservoirs were rapidly changing their trophic characteristics due to the addition of plant nutrients originating largely from human activities. The main nutrient sources identified were municipal and industrial wastewater and agricultural and urban runoffs.

Eutrophication is the response to this overenrichment by nutrients (primarily phosphorus and nitrogen) and can occur under natural or manmade conditions. "Manmade" eutrophication, in the absence of control measures, proceeds at an accelerated rate compared to the natural phenomenon. A recent survey (cf. Vollenweider 1979) has shown that eutrophication is one of the main forms of water pollution reported in countries throughout the world. The resultant increase in fertility in affected lakes, reservoirs, slow-flowing rivers and certain coastal waters causes symptoms such as algal blooms, heavy growth of certain rooted aquatic plants, algal mats, deoxygenation and, in some cases, unpleasant odor, which often affects most of the vital uses of the water, such as water supply, fisheries, recreation or aesthetics. In short, manmade eutrophication of inland bodies of water becomes synonymous with the deterioration of water quality and as such frequently causes considerable extra economic costs.

Manmade accelerated eutrophication can, in principle, be reversed by the elimination or reduction of the nutrient supply from such as municipal and industrial wastewaters, agricultural wastes and fertilizers. In most cases, however, it is not possible to eliminate all sources of nutrient supply. Thus, it is important to understand the qualitative and quantitative relationships which exist between nutrient supply and the degree of eutrophication in order to be able to develop sound lake management strategies to control eutrophication at minimum costs.

HISTORY OF OECD ACTIVITIES IN EUTROPHICATION

In 1967 a group of experts under the chairmanship of Professor O. Jaag (EAWAG, Zurich) recommended to the OECD that a comprehensive survey be made of the existing literature on eutrophication processes. This led to the publication of a report, "Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters with Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication" by Vollenweider (1968). This report introduced the concentration of nutrient loading and lake response but also stressed the inadequacy of limnological data for broad generalizations and for producing precise guidelines for eutrophication control.

Further, a symposium on "Eutrophication in Large Lakes and Impoundments" was held in Uppsala, Sweden, and the resulting report was published by the OECD in 1970.

In spite of the advances achieved in eutrophication control, many basic questions concerning eutrophication remained unanswered, and it became obvious that a broader limnological data base was required for intercomparison between bodies of water and assessment of the status of lake eutrophication. The nutrient loading concept and the related concept of loading tolerance had been consolidated and accepted by a large segment of the international scientific community, but controversies whether carbon and other growth factors rather than phosphorus or nitrogen limit algal growth in lakes continued for some time.

In 1971 the OECD established a Steering Group on Eutrophication and in February 1973, approved and adopted an "Agreed Program on Evaluation of Eutrophication Control" and charged the Steering Group on Eutrophication Control with the responsibility for developing and coordinating the agreed program, bringing into account its effectiveness, cost and feasibility. Four *ad hoc* expert groups carried out the program:

1. Expert Group on Detergents (published 1973);

2. Expert Group on Impact of Fertilizers and Agricultural Waste Products on the Quality of Waters (published 1973);

3. Expert Group on Wastewater Treatment Processes for Phosphorus and Nitrogen Removal (published 1974);

4. Planning Group on Measurements and Monitoring (published 1973).

The three expert groups and the planning group completed their reports in 1972. The planning report "Summary Report of the Agreed Monitoring Projects on Eutrophication of Waters" (published 1973) gave a common system of agreed measurements, guidelines on background data and comments on existing methods of sampling. It also outlined the basis for an international program of measurements and monitoring of waters being undertaken by interested OECD member countries. This program came to a closure in 1980 and has resulted in a Synthesis Report and four Regional Reports already being published. A fifth Test Case Report is presently in its final stage.

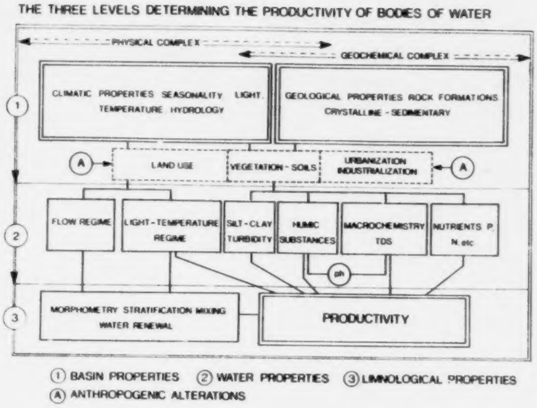
THE CONCEPTUAL BACKGROUND

Scientifically speaking, eutrophication is but a special aspect of water productivity. Seen in this perspective, studies on eutrophication have to respond to the same conceptual references as productivity studies in general. Productivity is the expression of the external physiographic complexes of the system as a whole, as well as of its internal physico-chemical and biological dynamics. Accordingly, the trophic properties of bodies of water, lakes, estuaries, sea coasts or running waters, have to be considered as the resultant of a sequential nexus of geographic, geochemical, climatic, hydrological and other factors.

In applying this concept to eutrophication studies on lakes and reservoirs, the scheme expressed in Figure 1 proposes *a quasi deductive procedure to derive the cause effect relationships which determine any observed specific limnological situation* from the characteristics of the catchment system by progression from the general properties of the system to the specific conditions of the water body considered. In the progression from level to level, the degree of freedom for the next level is narrowed down, i.e. the specific properties of the physico-geochemical complex at the top controls the hydrologic and qualitative properties and characteristics of the water deflux level, which in turn determines the limnological level in its connotation "productivity."

In order to bring this concept into perspective, at least one specific *transfer compartment* and two *transfer functions* have to be singled out:

A. The vegetation-soil complex acts as an intermediary between the physico-chemical complex and the water property level. Under natural conditions this compartment is practically the only source compartment in terms of nutrients, yet — due to man's intervention — has been substantially altered over the centuries. The historical and modern development in land use, urbanization and industrialization has had effects on both the size of this compartment in terms of nutrients potentially available, and on the transfer function, i.e. the amount of nutrients released per unit of time and unit of surface.


THE THREE LEVELS DETERMINING THE PRODUCTIVITY OF BODIES OF WATER
PHYSICAL COMPLEX
GEOCHEMICAL COMPLEX
CLIMATIC PROPERTIES SEASONALITY LIGHT, TEMPERATURE, HYDROLOGY
GEOLOGICAL PROPERTIES ROCK FORMATIONS CRYSTALLINE - SEDIMENTARY
①
A
LAND USE
VEGETATION - SOILS
URBANIZATION INDUSTRIALIZATION
A
②
FLOW REGIME
LIGHT - TEMPERATURE REGIME
SILT - CLAY TURBIDITY
HUMIC SUBSTANCES
MACROCHEMISTRY TDS
NUTRIENTS P. N. etc
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MORPHOMETRY STRATIFICATION MIXING WATER RENEWAL
PRODUCTIVITY
① BASIN PROPERTIES
② WATER PROPERTIES
③ LIMNOLOGICAL PROPERTIES
Ⓐ ANTHROPOGENIC ALTERATIONS

Figure 1. — Principal components and relationships determining the productivity of bodies of water.

The transfer function from the basin to the receiving waters is expressed in terms of *export coefficients* (e.g. kg/km^2 year) for each source. Point sources*, in general, are expressed in terms of *unit load*, yet in principle, they can also be expressed in terms of export coefficients at the condition that their density distribution can be established. The specific values of these export coefficients vary considerably from situation to situation, depending on the general geographic, climatological, hydrological and other conditions, as well as on the specific land use, urban and industrial development, etc. Export coefficients for phosphorus vary from less than 5 kg/km^2 y to over 500 kg/km^2 y and for nitrogen from less than 50 kg/km^2 y to more than 3000 kg/km^2 y.

In spite of this large variability, it is at times possible for specific geographic regions to apply lump values as has been shown by Vollenweider (1968, 1978) for average European conditions, and by Rast and Lee (1978) for U.S. conditions. However, uncritical transfer of such coefficients to unknown regions can lead to gross error.

B. *The nutrient loading concept*, as distinct from the transfer function refers to the receiving water body and *in most general form means the intensity of supply to a given body of water of any chemical factor necessary for plant growth*; in our context, however, its meaning has been restricted to nitrogen and phosphorus.

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* It is now customary to distinguish between diffused and point sources. However, such a distinction has primarily operational meaning: point sources, as opposed to diffused sources, in general, offer less difficulty for quantification, and at the same time are more amenable for technological control.

The theoretical limnology for decades has ignored this aspect, or at least neglected it, despite the early announcement made by e.g. Naumann (1932), Aberg and Rodhe (1942), Ohle (1955) a.o. Accelerated eutrophication of bodies of water over the last two or three decades has brought this problem into the open. The nutrient loading concept as defined here implies the connotation of a quantifiable property called "external load" which establishes the functional relationship between the basin and the trophic conditions of the receiving waterbody, and as such, is fundamental to the understanding of the total system.

From the methodological point of view, the quantification of the load-response relationship remains not without certain perplexities. Part of these relate to the question regarding the most appropriate way to express the load. Advantages and disadvantages of various options (e.g. absolute total amounts, specific loading per unit of surface or volume over a selected time-space, average inflow concentrations, etc.) are still a matter of discussion.

More important, however, the loading-trophic reaction relationship cannot be dealt with adequately without due consideration being given also to the fate of the various load components of a given substance within the lake system itself. An improvement over consideration of sole totals could be achieved by distinguishing at least two principal components and corresponding pathways, i.e. one component which enters the internal cycle via an "autotrophic" pathway — and which becomes immediately available to primary producers, and a component which enters into the internal cycle via a "heterotrophic" pathway of a more refractory nature (cf. Figure 2). In part, this aspect relates also to the question of what fraction is, or is not biologically available. In practice, the analytical distinction of these components is only partially possible, yet in order to understand the full array of reactions of different bodies of water to a given (total) load, a clarification of this problem is not without importance. Also, in many cases the internal loading cannot be neglected, though in many lakes this internal component remains far below the importance of the external loading. The exact quantification and dependency on external load is not entirely solved as yet, although essential progress has been made (cf. e.g. Golterman 1980).

However, important in this context is the basic idea that the in-lake bioproduction and recycling machinery (to use a more engineeristic analogy) is fed and driven by the external loading, and maintains itself depending on this external load in a repetitive cyclic steady state as long as no (unidirectional) alteration of the external supply occurs. On the other hand, any (unidirectional) change in the supply function will have as a consequence an alteration of the internal responses of the machinery speeding up or decreasing the velocities of exchange between the compartments, and correspondingly producing a change in size of each compartment.

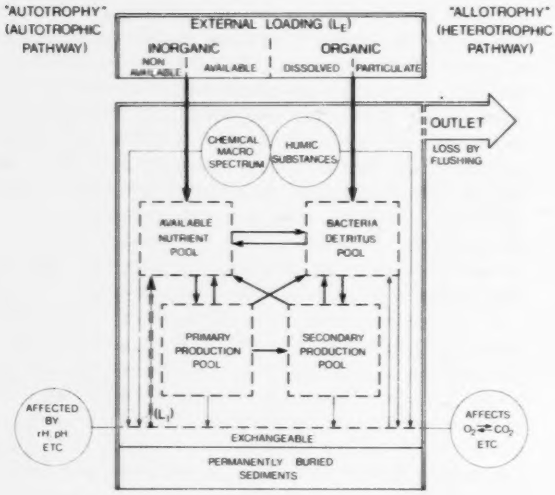

Figure 2. — Relationships between the principal lake-internal compartments and pathways of the external and internal loading components.

In pursuing this concept, the question is posed as to how far we can go at present to quantify the postulated relationships. This implies the necessity to establish and to elucidate the function of those parameters which primarily govern the relationship between the external load and the reaction of the body of water. From an applied point of view, such an understanding of the various relationships — always expressed in quantitative terms — would provide the scientific basis to develop criteria to manage the system; in particular, it would provide the basis to estimate the nutrient supply reduction required for lakes which in terms of preset water quality standards, appear to be over-fertilized.

The far reaching, practical, i.e. economical, implications of solving these questions have been recognized by OECD and have provided the motivations for the OECD Cooperative Program on Eutrophication which is the main theme of the following expose.

APPROACHES TAKEN IN THE OECD COOPERATIVE PROGRAM ON EUTROPHICATION

We realize that we have oversimplified the problem considerably, yet this has been done with the intention of bringing the problem into focus. Also, in speaking further on about the OECD Program, its scope, outcome and results, much oversimplification will be necessary, which does not exclude that the single collaborators, as well as the members of the Steering Committee, are well aware of the many difficulties arising in specific cases in applying a simplified approach.

To introduce the rationale for the OECD Cooperative Program on Eutrophication, it is necessary to recall the situation regarding the level of understanding of the nutrient load-trophic reaction relationship, particularly in regard to nitrogen and phosphorus, some 15 years ago. At that time, only a few reliable data on nitrogen and phosphorus loadings existed in the whole applied and theoretical limnological literature, and much of the data were no more than crude estimates which hardly permitted any founded generalization. Nonetheless, when the first author proposed in 1968 (cf. Vollenweider, OECD Technical Report) that in principle it was feasible to distinguish between "acceptable" and "excessive" loading, this proposal was welcomed in the scientific community, and immediately had substantial influence on practical decisions as well as stimulating a plethora of follow-up research.

It rapidly became clear through a number of meetings organized by OECD that only through international cooperation would it become possible to arrive at a sufficiently large amount of comparable data to derive valid quantitative relationships. Therefore, in about 1972 it was decided to launch a major cooperative program involving a majority of the OECD member countries. Some 18 countries, including more than 50 research centers covering between 100 to 200 lakes, have adhered to this program. It was conceived to tap into and make use of ongoing research but also to initiate new research. Accordingly, a full uniformity in approach could not expect to be achieved, yet this shortcoming was hoped to be counterbalanced by the large variety of individual lake situations covered by the program.

How did we develop this program? It was quite clear from the beginning that the focus would be on nitrogen and phosphorus, but that this aspect would have to be related to the particular geographic and limnological conditions of each lake individually studied. Further, it was necessary to develop a common language, to screen particular techniques and methods as to suitability and reliability, and to select those study items which appeared to be both pertinent to the success of the program, and logistically feasible, i.e. accessible for most cooperating centers involved. With evolving results, serious thought had to be given to data elaboration and exploration of the most useful way to correlate them.

In order to account for geographic variability, as well as for logistic considerations, we organized the program into four main projects:

1. An Alpine Project
2. A Northern Project
3. A Reservoir and Shallow Lake Project
4. A lump project for North America

Each project was headed by a regional coordination center, regional chairmen, plus some consultants forming a Technical Bureau for overall coordination. The first author has had the pleasure of chairing this committee over the last few years, and wishes to acknowledge the cooperation he enjoyed from his colleagues, particularly Drs. Ambuhl, Bernhardt, Forsberg, Golterman, Lee, Loffler, Maloney, Ravera and others, for steering this program, and the consultants responsible for synthesizing the material into report form (Drs. Kerekes, Clasen, Fricker, Lonholt, Ryding and Rast).

Table 1 provides an illustration of the kind of approach taken in developing a common language to identify parameters to be measured, or thought to be necessary to collate the information gathered into a consistent picture. It was understood that not necessarily all parameters would be measured in each individual case; some have been singled out as absolutely essential, whereas others have been left to the choice of the individual centers, in accordance with their capabilities and expertise.

In contrast to an approach of studying but a few examples only in depth, the chosen approach permits covering a wide spectrum of individual cases in an extensive way. Hence, our attention was not primarily focused on specific mechanisms, but on information that is amenable to *statistical analysis*. We wish to state this explicitly because at times the philosophy of the program has been misunderstood, particularly by those who expected a kind of material which could be used for dynamic modelling. From the very beginning, elaboration of the data at the basis of correlation and other comparative techniques thought to be meaningful, were envisaged. From this we expected to determine the cause-effect relationship in the sense of what we may call "statistical behavior," examples of which shall be given.

What kind of results have we obtained from this program? The program has covered a wide variety of limnological situations, including almost every type of lake and impoundment of the temperate region, and a few subtropical lakes and reservoirs, as well as some estuarine situations. Although the majority of lakes well studied fall into the meso- to eutrophic categories, a sufficient number of lakes representing oligo- and ultra-oligtrophic types have been included.

It is not the place here to discuss the whole array of results, conclusions and implications for practical management. These aspects are covered in the regional reports, in site specific reports and scientific papers, and in the final synthesis report. The integration of the available data has proven to be a worthwhile though difficult task. Such difficulties refer to both conceptual aspects as well as to straightforward problems with data screening, selection and appropriate interpretation. In many cases, it is not immediately available whether a data point represents a particular situation, a general uncertainty, or some unintentional mistake. Frequency of sampling, e.g., is a major factor in determining the reliability of a reported system itself. Superimposed on these problems are problems connected with calculation procedures; the choice of which of the various alternatives to use often remains a matter of taste rather than a matter of objective judgement.

As an example, loading figures represent a key parameter in the whole study, yet do we know little about the inherent uncertainty of any specific value reported. In our judgment, it is unlikely that individual year specific loading figures in most cases are better than ±25 percent. The natural year to year variability in loadings, in addition, is found to be in the same order

Table 1. — Categorization of parameters for measuring and monitoring eutrophication.

Ergodic (Resultant) Variables		Causative Variables
A. Short Term Variability: High	B. Short Term Variability: Moderate to Low	
- Phytoplankton biomass	- Zooplankton standing crop	- **Nutrient Loadings**
- Major algal groups and dominant species	- Bottom fauna standing crop	- Total Phosphorus
- Chlorophyll *a* and other phytopigments	- Epilimnetic Δ P, Δ N, Δ Si (Δ = difference between winter and summer concentrations)	- Ortho phosphates - Total Nitrogen
- Particulate organic carbon and nitrogen	- Hypolimnetic O_2 and Δ O_2	- Mineral Nitrogen ($NO_3 + NH_3$)
- Daily primary production rates	- Annual primary production	- Kjeldahl Nitrogen
- Secchi disk visibility		- **Nutrient Concentrations**
		- Same as above
		- Reactive Silica
		- Others (e.g. Microelements)

Related Descriptive Parameters

- Morphometric parameters of lake and catchment area
- Flushing regime
- Geological and climatic parameters
- Land use
- Urbanization and industrialization
- Main nutrient sources
- Temperature and mixing regime
- Conductivity, pH, alkalinity
- Major ion spectra
- Insolation and optical properties
- Others as deemed necessary

of magnitude (in some cases also considerably higher), so that representative loading estimates have a built-in uncertainty of at least ±35 percent. In-lake parameters such as biomass, chlorophyll, nutrient parameters, etc., are affected by similar uncertainties that have to be taken into account in data interpretation and correlation.

In its final ouput, the OECD Program has paid attention to the following aspects:

a. the qualitative assessment of the trophic state of bodies of water in terms of a few easily measurable parameters;

b. the dependence of this state on nutritional conditions and nutrient load;

c. translation of these results to the needs of eutrophication control for management.

One of the recurring problems we have run into during the study was the question of how to relate the classical trophic terminology — which is qualitative in nature — with the quantitative information provided in regard to selected parameters. In other words, the question arose of how far it is possible to quantify, in an objective way, the qualitatively defined trophic categories.*

Though apparently of academic interest, this question is not without meaning, in two ways. First, it relates to what has previously been stated relative to the need of a common language between limnologists themselves. Second, it relates to how the limnological terminology applies to practical management. From the practical point of view, there is no unequivocal relationship between the main trophic limnological categories and water usage. The relationship depends on specific use requirements. A categorization of bodies of water for fishery purposes need not necessarily correspond to the one for recreational purposes or to the one for domestic water supply, and none can entirely be matched with the limnological categories. Generally speaking, however, one can say that, proceeding from oligotrophy to eutrophy, multiple use of any water progressively becomes adversely affected with increasing trophy (cf. Table 2). Given this inherent ambiguity, therefore, it is important to attach quantitative meaning to the limnologically defined categories as the basic reference independent of their specific application. The OECD study has led to some interesting and not necessarily anticipated results.

The quantitative information given by the single contributors, together with their subjective judgment, were combined into a 4 x 5 matrix and for each block, mean and standard deviations have been calculated. A log-transformation of the original data was found to be necessary; the results are given in Table 3.

Table 2. — Trophic characterization of lakes impairment of various uses.

Limnological Characterization	Oligotrophic	Mesotrophic	Eutrophic
General level of production	low	medium	high
Biomass	low	medium	high
Green and/or blue-green algae fractions	low	variable	high
Hypolimnetic oxygen content	high	variable	low
Impairment of multi-purpose use of lake	little	variable	great

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* The pressing need for clarification in this context becomes apparent, if one recalls such examples as Lake Erie, which in the early sixties was "dead," then became "eutrophic," and finally is now considered, at least in regard to the main body of the lake, as mesotrophic.

Table 3. — Preliminary classification of trophic state in the OECD Eutrophication Program. Trophic status is assigned based on the opinion of the investigator of each lake. The geometric mean (based on log 10 transformation) was calculated after removing values < or > $\bar{x}$ 2 SD obtained (where applicable) in the first calculation.

Variable (Annual Mean Values)		Oligotrophic	Mesotrophic	Eutrophic	Hypereutrophic
Total	$\bar{x}$	8.0	26.7	84.4	
Phosphorus	$\bar{x} \pm 1$ SD	4.85-13.3	14.5-49	48 -189	
mg/m^3	$\bar{x} \pm 2$ SD	2.9 -22.1	7.9-90.8	16.8-424	
	Range	3.0 -17.7	10.9-95.6	16.2-386	750-1200
	n	21	19(21)	71(72)	2
Total	$\bar{x}$	661	753	1875	
Nitrogen	$\bar{x} \pm 1$ SD	371-1180	485-1170	861-4081	
mg/m^3	$\bar{x} \pm 2$ SD	208-2103	313-1816	395-8913	
	Range	307-1630	361-1387	393-6100	
	n	11	8	37(38)	
Chlorophyll *a*	$\bar{x}$	1.7	4.7	14.3	
mg/m^3	$\bar{x} \pm 1$ SD	.8-3.4	3. - 7.4	6.7-31	
	$\bar{x} \pm 2$ SD	.4-7.1	1.9-11.6	3.1-66	
	Range	0.3-4.5	3. -11	2.7-78	100-150
	n	22	16(17)	70(72)	2
Chlorophyll *a*	$\bar{x}$	4.2	16.1	42.6	
Peak Value	$\bar{x} \pm 1$ SD	2.6- 7.6	8.9-29	16.9-107	
mg/m^3	$\bar{x} \pm 2$ SD	1.5-13	4.9-52.5	6.7-270	
	Range	1.3-10.6	4.9-49.5	9.5-275	
	n	16	12	46	
Secchi	$\bar{x}$	9.9	4.2	2.45	
Depth m	$\bar{x} \pm 1$ SD	4.9-16.5	2.4- 7.4	1.5-4.0	
	$\bar{x} \pm 2$ SD	3.6-27.5	1.4-13	.9-6.7	
	Range	5.4-28.3	1.5- 8.1	.8-7.0	0.4-0.5
	n	13	20	70(72)	2

$\bar{x}$ = geometric mean
SD = standard deviation
( ) = value in bracket refers to the number of variables (n) employed in the first calculation

Clearly, most investigators consider a lake to be oligotrophic when the annual mean total phosphorus concentration is <10 mg P/m^3. It is noteworthy, however, that a few lakes with <10 mg P/m^3 were classified as either mesotrophic or eutrophic. Careful examination of the data revealed that in these cases the lakes have received an increased nutrient load in recent years, and as a consequence, have undergone some perturbation and change in trophic response. This may be in the form of a noticeable growth of attached filamentous algae along the shore near nutrient inflows, often accompanied by the appearance of a nuisance algae not observed before, however, without producing fundamental repercussions in the overall metabolism of the lake, noticeably its hypolimnetic oxygen conditions. At the other extreme, lakes with an annual total phosphorus concentration >30 mg P/m^3 and as high as 80 mg P/m^3 were assessed as mesotrophic by some investigators. In these cases, a variety of reasons, e.g. short water residence time or high turbidity, a high rate of grazing by zooplankton in the absence of fish, a.o., prevented the development of a high standing stock of phytoplankton, and hence, the lakes did not exhibit eutrophic charcteristics.

In regard to nitrogen, no consistent picture evolved. In particular, it was impossible to separate oligotrophic from mesotrophic lakes, although as a general rule, lakes of more eutrophic characteristics tend to have higher nitrogen concentrations.

A somewhat clearer delineation of trophic categories resulted, however, when allocation was based on chlorophyll *a* concentrations. In general, lakes were assessed as oligotrophic, mesotrophic or eutrophic when annual mean chlorophyll *a* concentrations were <2.5 to 10, or > 10 mg chl *a*/m^3, respectively. No lake was classified as eutrophic with an annual mean concentration of chlorophyll *a* < 2 mg/m^3. In regard to "worst case" situations, i.e. peak chlorophyll values, lakes are considered to be oligotrophic, mesotrophic and eutrophic when annual peak chlorophyll *a* concentrations are around 5, 16 and > 25 mg/m^3, respectively.

What emerged from the assessment of all information available, however, led to the conclusion that there is no possibility of defining strict boundary values between trophic categories. While the progression from oligo- to eutrophy is a gliding one — as has been stressed many times in the past — any one combination of trophic factors, in terms of trophic category allocation, can only be used in a probabilistic sense. The probability distribution for the two single factors, yearly average phosphorus and chlorophyll, for the three main categories (oligo, meso, eutrophy) plus the two boundary categories (ultra-oligo and hypertrophic) is exemplified in Figures 3 and 4, e.g. the probability of classification of a body of water having a total phosphorus concentration of 10 mg/m³, respectively, would be as follows:

	Phosphorus	Chlorophyll
ultra-oligotrophic	10%	6%
oligotrophic	63%	49%
mestrophic	26%	42%
eutrophic	1%	3%
hypertrophic	0%	0%

In judgment terms, then, such a water body is best classified as oligotrophic with a certain tendency toward mesotrophy. However, exceptionally, such a body of water may have excellent ultra-oligotrophic characteristics, or to the contrary, may show signs of grave deterioration, as is the case with Lake Mjosa.

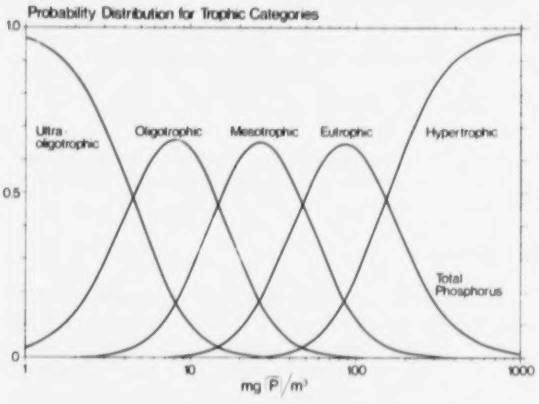

Figure 3. — Probability distribution of trophic categories relative to average phosphorus concentrations.

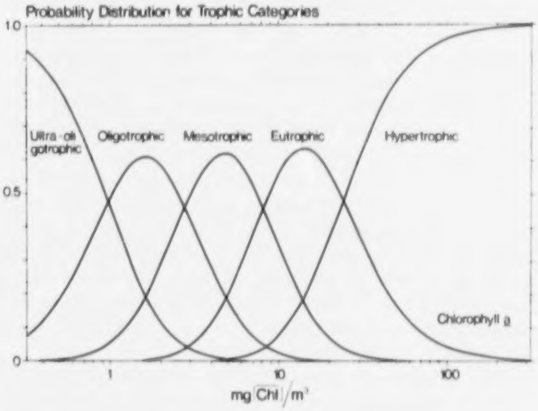

Figure 4. — Probability distribution of trophic categories relative to yearly average chlorophyll concentrations.

Evidently, this way of looking at trophic ca[illegible]ation has considerable management implicat[illegible], in a given case (e.g. a drinking water reservoir), it is important that certain water quality characteristics are maintained, then the management objectives must be set at some level slightly lower than would be required for maintaining average conditions.

To manage a lake with a certain objective in mind, we need knowledge of (i) which of the nutritional factors controls the system and (ii) what the relationship is between nutrient loading and the trophic reaction of the lake.

In regard to the first aspect, one of the primary results evolving from the data collected is confirmation that in at least 80 percent of the cases studied phosphorus was found to be the production-controlling factor; some cases remained inconclusive and the rest were identified as nitrogen limited, or controlled by some other factor.

In regard to the second aspect, the relationships between trophic characteristics, such as nutrient concentrations, mean annual chlorophyll, peak chlorophyll and loading, have been shown to be amenable to quantification; in accordance with the program objectives, these relationships are statistically not deterministically defined. Hence, if these relationships are to be used for prediction, the built-in uncertainty has to be taken appropriately into account.

Restricting the discussion to phosphorus, the results are based on the following methodology:

It has been obvious for some time now that simple relationship between areal or volumetric loading and lake phosphorus levels cannot be established without consideration of sedimentation and flushing (cf. e.g. Vollenweider 1969, 1975, 1976; Dillon 1975, Kerekes 1975). Basically, this relationship has to be thought of as follows:

In the most simple way, this scheme can be expressed mathematically as

$$\frac{d[\bar{P}]_\lambda}{dt} = \left(\frac{1}{\bar{\tau}_w}\right)[\bar{P}] - \left(\frac{1}{\bar{\tau}_p}\right)[\bar{P}]_\lambda \qquad \text{Eq. 1}$$

where

$[\bar{P}]_\lambda$ = average total lake concentration (which includes both dissolved and particulate phosphorus components)

$[\bar{P}]_\lambda$ = average inflow concentration of total phosphorus

$\bar{\tau}_p$ = average residence time of phosphorus

$\bar{\tau}_w$ = average residence time of water

The righthand terms represent the average rate of supply to and the average rate of loss of total phosphorus from the lake, respectively, and the lefthand terms the corresponding temporal variations of the average lake concentration. Note that in this formulation no specific assumptions are made as to the mechanism of loss.

Several possibilities are open to deal with the above equation 1, yet in principle, it reduces to evaluating statistically the quotient $\bar{\tau}_p/\bar{\tau}_W$ as a function of parameters controlling the system such as mean depth, epi-hypolimnion ratio, hydraulic load, length of stratification, etc., assuming steady state conditions.*

From the various attempts made to analyse these relationships, the fact evolved that mean depth and hydraulic load are the most important factors, and that $\bar{\tau}_p/\bar{\tau}_W$ can be approximated by a function of the form

$$\bar{\tau}_p/\bar{\tau}_W = [\bar{P}]_\lambda/[\bar{P}]_\lambda \approx \frac{1}{1 + a \cdot q_s^b \bar{Z}^c} \qquad \text{Eq. 2}$$

Approximate values for *a*, *b* and *c* were found to be 1, -.5 and +.5 so that equation 2 reduces to

$$\bar{\tau}_p/\bar{\tau}_W = [\bar{P}]_\lambda/[\bar{P}]_, \approx \frac{1}{1 + \sqrt{\bar{z}/q_s}} = \frac{1}{1 + \sqrt{\bar{\tau}_W}}$$

(Vollenweider 1976). These findings correspond to results of similar approaches made by Larsen and Mercier (1975), Dillon (1974), Kirchner and Dillon (1975), Chapra (1975), Chapra and Tarapchak (1976), Reckhow (1978), a.o. which all are variations of the same theme.

Accordingly, mean lake phosphorus should be predictable from load, in principle, by

$$[\bar{P}]_\lambda = [\bar{P}]_, / (1 + \sqrt{\bar{\tau}_W})$$

Figure 5 shows that this indeed is the case yet (2b) slightly underestimates concentrations at low levels, and overestimates concentrations at higher levels.

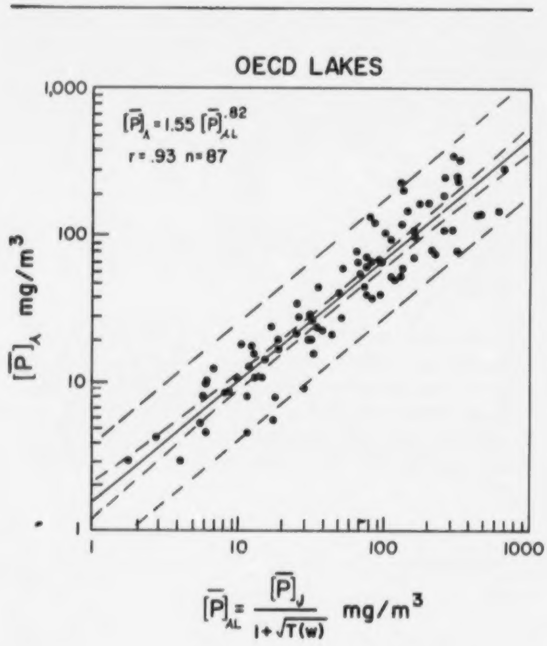

Figure 5. — Relationship between flushing corrected average inflow concentrations and average lake concentrations of phosphorus.

All these formulations, and their variations, contain the underlying assumption that lakes can be treated as mixed reactors in steady state. This is not true for most lakes. It is therefore surprising that simplified relationships of this sort provide a workable basis, which in principle means that a large spectrum of lakes (governed by phosphorus) behave statistically in a similar way. *The relationships derived describe the average statistical relation pattern of lakes between phosphorus load and phosphorus concentration.*

Used as a diagnostic criterion, these relationships also provide a tool to identify "outlayers." The term "outlayer," as used here, refers to both statistical and functional variability. Indeed, outlayers from the rule may indicate simple data uncertainty as Rast and Lee (1978) have shown to be the case for lakes for which the load has been either under- or over-estimated. However, outlayer lakes have also been identified which behave functionally differently; either the assumption of steady state does not apply, their sedimentation quota is above or below normal, or an internal or external disturbance of the system exists. Examples for each possibility could be listed, yet more importantly, in most cases it was possible to identify the reason for deviation.

This experience shows that it would be wrong to discard equation I simply because a given data set would not fit it. *Strong deviations* from this relationship can be used as a *diagnostic indication* for a particular situation which requires further attention. Conversely, it would be wrong to blindly apply this relationship for predictive purposes, regardless of special limnological conditions.

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*The term "steady state" is referred to in this context as "repetitive state over time" for which $\Sigma \pm d[\bar{P}]/dt = 0$. Time resolution is 1 year.

The next step in the sequence was to establish the relationship between chlorophyll (yearly and peak values) and nutrient concentrations. Without entering into detail for the present review, it may be said that, on average, the yearly mean chlorophyll concentration was found to be between 25 to 30 percent of the average total phosphorus (cf. Figure 6A). Peak chlorophyll values (which are of particular importance for practical considerations) on the other hand, resulted as roughly three times average chlorophyll, but exceptionally can be considerably higher (cf. Figure 6B).

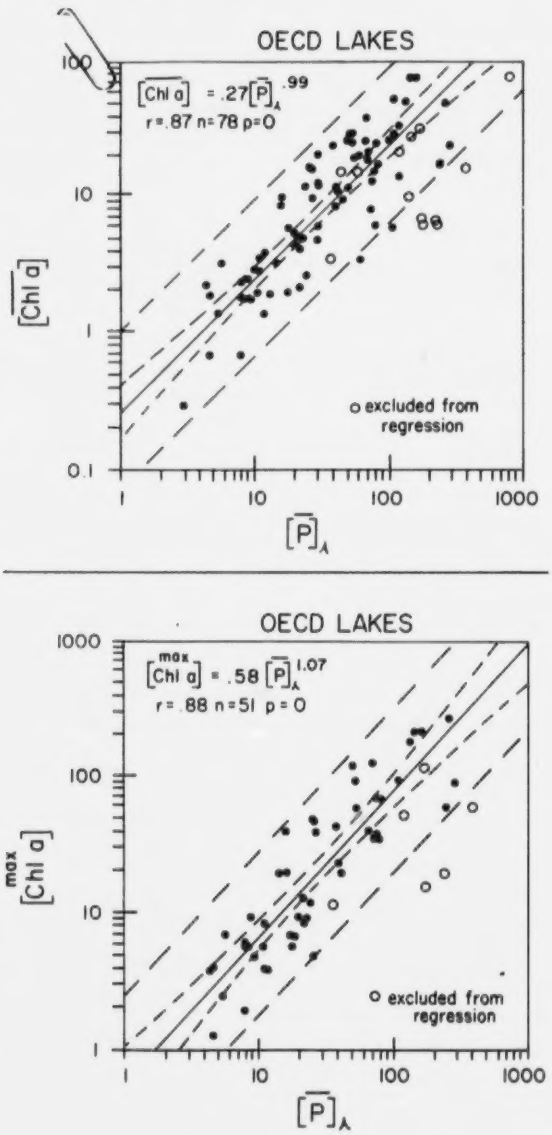

Figure 6. — Relationship between average lake phosphorus concentrations and chlorophyll. *A*. Yearly average chlorophyll. *B*. Peak chlorophyll observed.

Interestingly, chlorophyll apparently also resulted in being correlated to nitrogen in many cases, yet statistical discrimination tests have shown that this is primarily due to coupling of nitrogen with phosphorus. In particular cases, however, the dependence of bioproductivity on nitrogen, as well as on other factors, has been found to be unquestionable. The interaction between phosphorus and nitrogen has been identified as an area which requires further research.

In the light of what has been said thus far, a close relationship between phytoplankton biomass (as measured by chlorophyll) and phosphorus load can be expected. The findings are illustrated with Figures 7A and 7B. In regard to statistical variability, the same applies regarding the relationship between phosphorus loading and concentration. However, it is to be stressed that chlorophyll is but a crude parameter to estimate biomass. Indeed, cases did come to light indicating that the biomass/chlorophyll ratio can vary by a factor of up to 3 and is therefore a major contributor to the scattering observed.

Nevertheless, the biomass (chlorophyll)/loading relationships are perhaps the most important results of the OECD study thus far. Within the range of the identified uncertainties, they permit estimation of the phosphorus reduction necessary to reduce eutrophication to any preset level of biomass. The main conclusion which one can draw from the OECD results is the fact that the production level of any given water body, in principle, is proportioned to its nutrient load, and therefore that load reduction will have effects proportional to the reduction achieved.

In the long run and with consideration that exceptions from this rule exist, it is desirable to base such judgments not solely on standing crop but also on related dynamic parameters. Unfortunately, the OECD study has not permitted convincing establishment of relationships between loading and dynamic parameters, such as primary production and hypolimnetic oxygen depletion rates, etc. This is due, in part at least, to the dearth of usable data points and in part to considerable difficulties in measuring such parameters uniformly. The problem of hypolimnetic oxygen conditions is further compounded by conceptual uncertainties (e.g. oxygen depletion rates versus apparent or potential oxygen deficit).

The following is a short account of the present state of the art.

The relationship between primary production and phosphorus deviates structurally from the chlorophyll - $[\overline{P}]$ relationship by its non-linearity. This is due to the self-shading effect of the biomass with increasing levels of productivity which can be dealt with by introducing a generalized primary production model. This model assumes that the annual primary production can be expressed with a hyperbolic function similar to that of a daily photosynthesis integral (cf. Vollenweider 1970) i.e.

$$\Sigma C\ (g/m^2 \cdot y) = K \cdot \frac{[\overline{chl}]}{\varepsilon_w + \eta\,[\overline{chl}]}$$

where [chl] is the average yearly chloropohyll concentration of the euphotic zone, ε_w a characteristic average extinction coefficient (1/m) which includes turbidity, humic substances and other colored substances, and η the specific vertical extinction coefficient per unit of chlorophyll.

In order to establish the relationship to nutritional conditions, the chlorophyll term in this equation can be substituted by the corresponding relationships, and the remaining parameters calculated from measured data by least square techniques. Correspondingly, the hypolimnetic oxygen depletion rates should be predictable from primary production. This hypothesis further implies that the relationship between oxygen depletion rates (expressed as areal hypolimnetic oxygen depletion rates) and nutritional conditions, should parallel those for primary production.

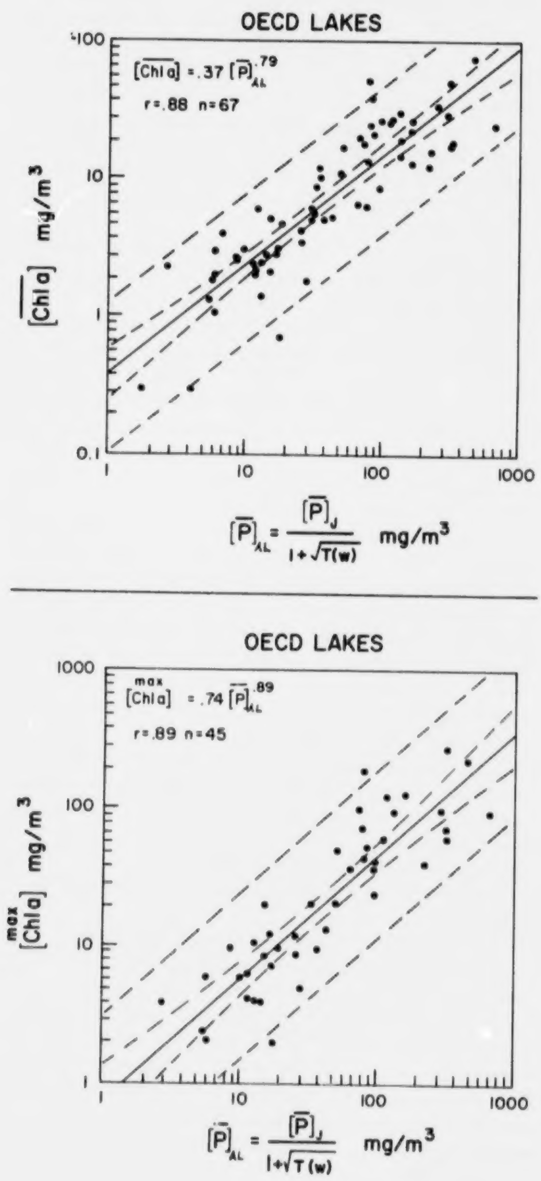

Figure 7. — Relationship between flushing corrected average inflow concentrations and chlorophyll. *A*. Yearly average cholorophyll. *B*. Peak chlorophyll observed.

Our preliminary results show that this is indeed the case. Yet, the much larger scattering of the data points also shows that factors other than those taken into account in our analysis are involved in determining primary production and hypolimnetic oxygen conditions. The higher uncertainty, e.g., in linking hypolimnetic oxygen depletion rates with loadings, as found in our study, depends undoubtedly on the complex interactions between the epilimnetic and hypolimnetic regime in each individual case. The underlying factors relate to specific lake morphology, length and type of thermal stratification, vertical entrainment and oxygen transfer, and interactions between sediments and overlying waters. It is evident that, in order to reduce the uncertainties, much additional work is required.

HOW FAR DID THESE PRELIMINARY RESULTS MEET OUR EXPECTATIONS?

Considering the large variety of lakes examined, and considering also the unavoidable inequality in the data collected, the results achieved to date probably exceed by far what could be expected from this program. Admittedly, some of the correlations of factors thought to be interrelated, in part, were found to be poor, yet at least some of the more important correlations turned out to be highly significant (cf. relationship between phosphorus load and in-lake phosphorus concentrations, between this latter and chlorophyll, and between loading and chlorophyll).

Generally speaking, what has been achieved in terms of understanding lake behavior, lies say, half way between the historic position that each lake is an entity which has to be understood on its own, and solely on its own, and an advanced but not yet attained level of insight which would make it possible to deduce the reaction of bodies of water with a high degree of precision from a few parameters.

The program, seen in its totality, has provided a unique opportunity to study limnology in a comparative sense. In this respect, it can be considered as a milestone in national and international cooperation, the prospects of which are manyfold and leading into the direction of what Elster outlined as the future of limnology in his memorable 1956 conference (cf. Elster 1958).

However, the program would have failed if it had not also provided the basis from which it is possible to establish improved loading criteria for practically combating eutrophication of lakes. A synthesis of such criteria is given in Figure 8. These criteria are in logical sequence of the criteria proposed in previous papers by

Vollenweider (1968, 1975, 1976), linking average inflow concentrations for phosphorus with expected average lake concentration and average chlorophyll concentrations as a function of the flushing regime of lakes. Division between the main trophic categories is based on the 50 percent probability of belonging to the indicated class, and the vertical arrows may be read in the sense of "belonging to or better as" the indicated class. With this, management has a tool to establish whatever goal is thought to be desirable to reach, or conversely, to anticipate the level of improvement which can be expected from an established reduction program. How this should be done in practice, and with what level of uncertainty one has to reckon with, is discussed in the Synthesis Report.

Besides this positive note, however, it must be underlined that many questions remain open, and that a blind and uncritical application of the OECD results can lead to gross error. Limnology, and its application for practical purposes, was and is a complex science, and remains a matter of skill and experience. The establishment of group behavior of lakes, as was the main objective of the OECD Program, does not necessarily mean that each single case can be subordinated to one single rule.

Indeed, a more detailed elaboration of the OECD data — a work which still requires considerable time — already indicates that a more selective grouping of lakes having similar limnological properties would reduce some of the uncertainties resulting from an indiscriminate pooling of all data. From here on, one has to find out what the discriminative parameters are for group differences. Factors which lend themselves for further consideration are: type and length of stratification, epi- hypolimnetic ratio, mixing depth, ice coverage, humic substances, N/P ratio, zooplankton and fish population, etc.

An improved approach to discrimination analysis of trophic conditions of lakes is underway by Chapra and Reckhow (1979) who try to avoid some of the pitfalls of the hitherto used prediction models by applying the uncertainty theory. Schaffner and Oglesby (1978) and Oglesby and Schaffner (1978) introduce in their modifications some of the factors mentioned.

Last but not least, the next step in the endeavour will be a concerted effort to link experimental with theoretical limnology. Over the last decade or so, theoretical limnologists have made much progress and brought into the open many of the uncertainties in our understanding. This throws the ball back to experimental limnologists who will have to rethink many of their programs. The extended experimental work of Schindler and his colleagues in the Experimental Lakes Area studies — which cannot be referred to in detail in this review — provides further guides to understanding the complex relationship between nutrient loading and lake reaction.

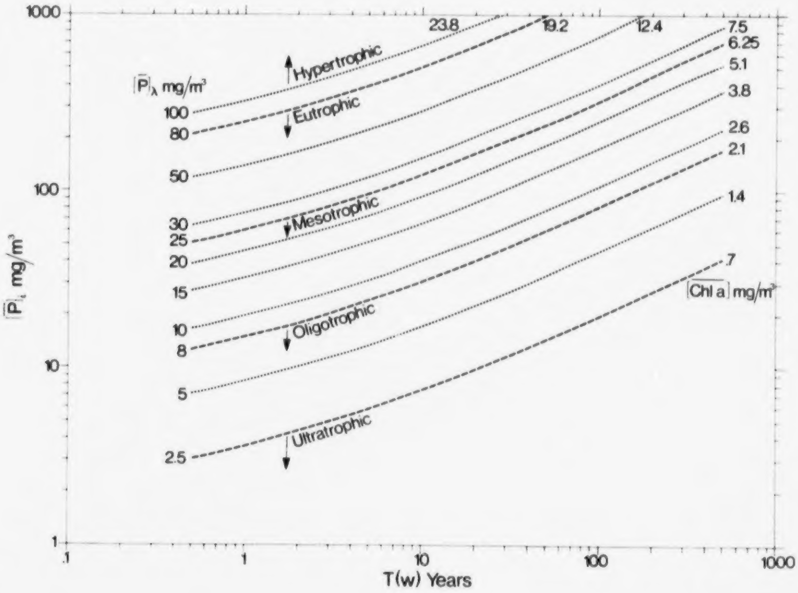

Figure 8. — Synthesis of the OECD information: Group relationships between average inflow concentrations and average lake concentrations of phosphorus, average yearly chlorophyll concentrations, and trophic categories relative to the water residence time of lakes.

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The OECD Cooperative Program on Eutrophication would not have been possible without the efforts and generous contributions made by all collaborators of this program. It is impossible to list names individually. However, as principal author of this paper and Chairman of the Technical Bureau, I wish to express my thanks and those of the members of the Technical Bureau to all colleagues, advisers, helpers, governmental and other agencies, who made this unique collaborative program possible.

PRESENT KNOWLEDGE OF LIMITING NUTRIENTS

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ABSTRACT

To develop an effective control program, it is important to know which factor limits maximum biomass. Because peak algal mass often appears during a short period of time, studying the standing crop limiting nutrient is probably more convenient than developing a water sampling program to cover irregular peak situations. Nitrogen, phosphorus, and chlorophyll *a* are discussed.

For developing effective control programs it is important to know which factor controls or limits the maximum biomass developed during, for example, the summer period. Analyzing the limiting nutrients during a limited period of time, e.g., 1, 2, or 3 weeks, when the phytoplankton development is rapid and the absolute concentrations of available nutrients change rapidly, can be looked upon as a study of production or productivity limiting nutrients. The evaluation of the limiting roles of, for example, nitrogen and phosphorus, is then based on the amounts of the available forms, No_3-N_1, NH_4-N, and PO_4-P. If these are present in excess they can't limit the biomass development. To be considered limiting the concentrations have to be close to zero.

As peak algal mass often appears during a short period of time it can be difficult to have a water sampling program covering just this situation. Peaks can also appear irregularly from year to year, depending on climatic conditions. Therefore, methods analyzing limiting nutrients during a short and chemically-biologically intensive period of time may be difficult to use for lake water management. Included here must also be the problems with "luxury" consumption. Algae can, as is well known, assimilate and store P for later use. This means that in spite of PO_4-P concentrations in the surrounding water being close to zero, phosphorus may not be the limiting nutrient.

For water management, it is probably more convenient to study the standing crop limiting nutrient. This concept can be developed by looking at nutrient and biomass levels without taking special notice about assimilable forms of nutrients or the physiological processes developing this specific biomass level. The amounts of total-N and total-P and the ratios of N to P in relation to, for example, chlorophyll or transparency can be studied. This approach is easier to handle for water management.

Comprehensive results demonstrate linear correlation between summer averages for total-P and chlorophyll *a* up to a P concentration level corresponding to 100 mg/m³. Below this concentration level, P can be considering as limiting phytoplankton standing crop. Above, nitrogen will take over in relation to phosphorus. N_2-fixing blue-greens will often help a nitrogen stressed situation, which means that nitrogen has less chance to act as a limiting nutrient. As a guide for indicating their roles, the following ratios of N to P can be used:

Total-N / Total-P	Limiting nutrient	Chlorophyll *a* level, mg/m³
> 17	P	< 20
10 — 17	N and/or P	20-70
< 10	N	>70

These values are based on summer average values. As recent studies demonstrate very strong correlation between summer average and summer maximum values, reliable average values will also give good information of the worst situation; this knowledge is essential for water management and physical planning.

NON-NUTRIENT FACTORS INFLUENCING THE DYNAMICS OF EUTROPHICATION

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ABSTRACT

Annual primary production — chosen as trophic state index of a lake — is determined by the following non-nutrient factors: Morphology, hydraulics, meteorology, internal mixing, non-nutrient water chemistry, and input of inorganic particles. As a consequence of various anthropogenic activities, in many lakes some of these factors are influenced by man. Internal mixing processes are considered the key to understanding natural and artificial non-nutrient eutrophication factors since they control the vertical transport of nutrients from the sediments into the hypolimnion and across the thermocline of lakes. During the stratification period, vertical mixing — although slow but quasi-continuous — may cause an enormous internal loading to the trophic zone of shallow or medium deep eutrophic lakes. In winter, single meteorological events can be responsible for the intensity of deep water renewal and thus for the chemical dynamics in the hypolimnion during the subsequent stagnation period. Minor physical perturbations can significantly change an existing mixing pattern and thus the trophic evolution of the lake. On the other hand, physical alterations may intentionally be combined with external measures to control lake eutrophication. However, this will not be unproblematical unless a better understanding of the relation between external physical forces and internal mixing processes has been achieved.

Annual primary production — chosen to measure the dynamics of eutrophication in lakes — is determined by various external and internal factors among which the input of nutrients has been identified as the most important cause of eutrophication. However, because of non-nutrient factors lakes vary greatly in their response to increasing nutrient input. Non-nutrient factors are:

1. Morphology (lake size, mean depth, shape of lake basin)
2. Hydraulics (water residence time, type of inlets and outlets)
3. Meteorology (solar radiation, temperature, wind)
4. Internal mixing
5. Non-nutrient water chemistry
6. Input of particles and sedimentation patterns.

Analyzing lakes from different climatic zones of the earth, Brylinsky and Mann (1973) have found that the input of solar radiation is the most important factor regulating primary productivity of a lake. For the more homogeneous climate zones of Central Europe and North America, where the problem of lake eutrophication is most urgent, nutrient input represents the dominant influence on trophic state.

As a consequence of various anthropogenic activities, in many lakes not only nutrient input, but also some of the non-nutrient factors, are (intentionally or inadvertently) influenced by man. This is mainly the case with respect to hydraulic properties of lakes (for instance, flood control by diverting rivers through natural lakes or by dams) whereas morphology and meterology are still beyond man's technical ability (at least for larger lakes). Non-nutrient water chemistry, especially the input of salt, may influence the internal mixing properties of a lake by affecting density. Particle loading affects the transparency of the water column giving rise to a different structure of primary production and vertical temperature distribution because of different adsorption of solar radiation.

Most of these phenomena affect directly or indirectly the internal mixing processes which are considered to be the key to understanding natural and artificial non-nutrient eutrophication factors. Internal mixing controls the vertical transport of nutrients from the sediments into the hypolimnion and across the thermocline (internal loading). A growing tendency exists for man to use lakes in ways which change their mixing pattern. Examples are the use of lakes for hydropower and irrigation, the input of waste heat, the export of heat for heat pumps, and the use of natural lakes for pumped storage power operation (Imboden 1979, 1980). Physical alterations may also intentionally be applied in combination with external measures to control lake eutrophication (artificial mixing, hypolimnion drainage).

Recently, Imboden and Gachter (1979) have extensively discussed the impact of physical processes on the dynamics of eutrophication. Since in most lakes phosphorus has been found to be the controlling input, they analyze the relationship between annual P-loading (Lp) and primary productivity ($\bar{z}$: P) to identify factors other than P-loading which influence productivity.

The data used for this analysis (Figure 1) originates from lakes of the relatively homogeneous climate of Europe and North America where the dominant influence of solar radiation mentioned earlier is less important.

Another factor has been brought forth by Vollenweider (1968), who found an increase of nutrient tolerance with increasing lake depth. Indeed, the data in Figure 1 show a slightly higher productivity for shallow lakes, a tendency mainly associated with eutrophic lakes. A multidimensional regression analysis results in the equation (z: mean depth)

$$\log \Sigma P = 2.6 - 0.24 \log \bar{z} + 0.66 \log L_P \qquad \text{eq. 1}$$

(correlation coefficient: 0.76)
but again the scattering is far too large to be acceptable for a unique theory of primary productivity.

The existence of an L_P-dependent lower limit for ΣP, a salient feature of Figure 1, suggests that ΣP be divided into the two components

$$\Sigma P = \Sigma P^0 + \Sigma P^1 \qquad \text{eq. 2}$$

ΣP is the productivity in a hypothetical isolated system having the size of the trophogenic layer, i.e., a system which is neither in contact with the sediments nor influenced by turbulent nutrient transport across the thermocline. It can be approximated by a simple steady-state one-box model for primary production (see Imboden and Gachter (1979) for details).

P^1 is the contribution from internal nutrient recirculation. It is this part of ΣP which is expected to be sensitive to the mixing regime of the lake, its morphometry and redox conditions at the sediment-water interface.

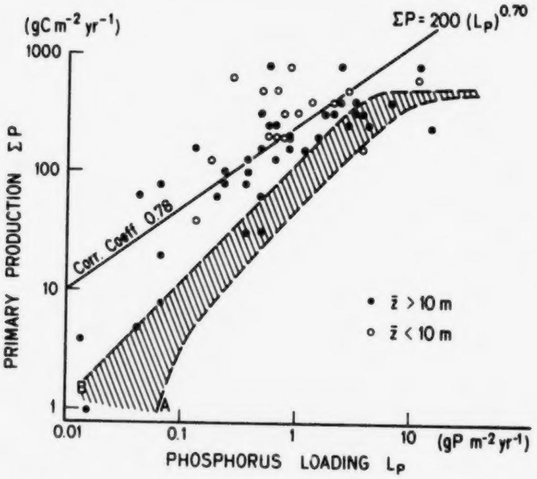

Figure 1. — Correlation between annual P-loading per surface area(L_P) and primary productivity(ΣP) for various European and North American lakes. Shaded area: Results from steady-state productivity model for epilimnic residence times τ_E of water between 0.06 yr (curve A) and 15 yr (curve B). Shallow lakes (mean depth $\bar{z}$<10 m) show a slight tendency toward higher productivities (from Imboden and Gachter, 1979).

Imboden and Gachter summarize their analysis of primary productivity data as follows:

1. Productivity mainly depends on P-input, but there exists no unique connection between ΣP and L_P neither in the measured data nor from theoretical considerations.

2. The minimum productivity ΣP^0 is reasonably approximated by a simple steady-state one-box model.

3. At high P-loading, the same model also reproduces the saturation effect on P around 400 g C m^{-2} yr^{-1}.

4. The "internal fraction" ΣP^1, i.e., the difference between the measured productivities and the corresponding ΣP^0, is largest for medium L_P values (around 1 g P m^{-2} yr^{-1}).

5. P^1 exhibits a tendency to be larger for shallow lakes, indicating the role of internal mixing. However, mean depth alone cannot explain the magnitude of P^1; probably other factors such as morphometry of the lake, exposure to wind, vertical mixing intensity and the redox conditions at the sediment-water interface are of equal importance.

6. As exemplified by the curves A and B in Figure 1, hydraulic loading is only of limited influence on P.

The statistical approach provides some basis for speculation on the possible influence of vertical mixing on the trophic state, but our present knowledge of lake mixing mechanisms does not permit quantifying these effects in a general way. The mechanisms of how kinetic energy, entering the lake by sheer forces of wind stress and by rivers, is transformed into turbulence and finally dissipated into heat by viscosity is not fully understood. Lakes can be even less classified by simple schemes with respect to their physical characteristics and biological phenomena.

At this time, case studies are most suitable to reveal the physical processes which may interfere with trophic conditions. In the highly eutrophic Greifensee (Switzerland), Imboden and Emerson (1978) have determined the vertical eddy diffusivity from the distribution of radon-222, a natural radioactive isotope, in the water column. Together with measured vertical phosphorus gradients, they estimate internal loading in this lake to be between 60 (June to September) and 100 percent (October to November) relative to external P input.

The following general statements can be derived from the case study of Greifensee:

1. Mineralization of organic material in the hypolimnion and at the sediment-water interface during stagnation leads to vertical concentration differences between epilimnion and hypolimnion. Qualitatively, the gradients are inversely related to the hypolimnic/epilimnic volume ratio, i.e., shallow lakes generally have larger gradients.

2. In the case of phosphate, the mean depth dependency of vertical gradients is further enhanced by the factor of hypolimnic O_2 depletion. Under anaerobic conditions released phosphate at the sediment surface is not bound by surface absorption mechanisms. The occurrence of anaerobic conditions is more probable in shallow lakes since, given a certain production of biomass at the surface, it is the size of the hypolimnic oxygen reserves (i.e., of the hypolimnic volume) which determines at what time of the summer anaerobic conditions occur.

3. As a consequence of (1) and (2), the trophic level of shallow lakes should show a larger sensitivity with respect to external nutrient input than deep lakes, which is in accordance with the early findings by Vollenweider (1968).

4. One effect partially counteracting these above statements consists in the hidden correlation between lake mean depth and surface area. Large lakes (which

are often the deeper ones) generally have a higher vertical mixing intensity which would favor transport from the sediments through the thermocline into the trophogenic layer. However, this effect seems to be less pronounced than the other ones.

As mentioned before, the influence of the mixing processes on primary productivity cannot, in general, be quantified. But it is possible, at least, to predict whether productivity would increase or decrease with vertical mixing intensity.

In Table 1, a summary of the relevant mechanisms and their sensitivity with respect to vertical mixing is given. In most cases, only a careful weighing of opposite effects is necessary to predict the behavior of the system, a task possible only by using numerical lake mixing models. A treatment of these models lies beyond the intention and possibility of this contribution.

Table 1. — A summary of mixing processes and their influence on primary productivity.

Under the influence of the following changes, primary productivity would	
Increase	Decrease
A. Deepening of the thermocline	
Higher pool of nutrients	Lower rate constant for productivity due to lower temperature
	Less favorable ratio between zone of respiration and production
B. Increase of vertical mixing in the thermocline and the hypolimnion	
Increase of internal nutrient loading	Decrease of nutrient flux from the sediments since the sediment surface remains aerobic during a longer period
	Decrease of biomass density by dilution due to mixing
C. Increase of the probability that the lake undergoes total turn-over during winter (of importance only for meromictic lakes)	
Recycling of hypolimnic nutrient pool leading to larger initial concentrations in spring	Decrease of sediment boundary flux due to higher hypolimnic oxygen concentrations
Turbulence decreases nutrient retention (lower sedimentation velocity and/or resuspension of sediments)	

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DYNAMICS OF NUTRIENT ENRICHMENT IN LARGE LAKES: THE LAKE MICHIGAN CASE

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ABSTRACT

Lake Michigan is an interesting case to consider in discussing the dynamics of eutrophication. Although many environmental changes indicate that accelerated eutrophication has occurred, the main body of the lake is primarily mesotrophic. For example, open lake chlorophyll *a* concentrations do not exceed 3 to 4 μg/liter during the spring bloom and concentrations on a lake-wide basis average 8 μg P/liter. Results of nutrient enrichment experiments with natural phytoplankton assemblages show that phytoplankton growth can be increased with small phosphorus additions and that effects of phosphorus are greater when water is enriched simultaneously with phosphorus and trace constituents (EDTA, trace metals, and vitamins). Effects of eutrophication are manifested to the greatest extent in nearshore areas where, if localized nutrient sources are adequate, phytoplankton standing crops may be many times greater than in the offshore waters. Data on nutrient inputs show that the effects should be localized because inputs are not uniform over the lake basin.

INTRODUCTION

The dynamics of eutrophication in large lakes must be considered from a different perspective than that for small lakes. Data for Lake Michigan will be used to illustrate the point that size is an important consideration. Lake Michigan is a large lake, having a water surface area of 56,500 km^2 and a water volume of 4,800 km^3 (Table 1). The main axis runs north-south from 42-46°N so one would expect latitudinal influences on lake processes. The main outflow is to the north through the Straits of Mackinac and the main sources for nutrient loading are in the southern part of the lake. This means, therefore, that nutrients must be transported from south to north to be removed from the lake with the outflow.

The lake also may be divided into a nearshore zone and an offshore zone. The water quality of the nearshore zone, as will be shown in this paper, is very distinct from that of the offshore waters. The nearshore zone differs from the offshore zone in that it receives higher nutrient loading from tributaries. In addition, physical processes in the nearshore are distinct from the offshore, currents are stronger, and effects of waves and currents, particularly relative to interactions with the sediments, are greater.

In this paper the nearshore zone has been defined arbitrarily as that area lying within the 30-meter contour line. This is roughly the average coastal or nearshore zone suggested by Mortimer (1975) who stated that the nearshore strip, about 10 kilometers wide, is the "scene of the main transfer of energy from wind to total basin motion and contains the greater part of the lake's kinetic energy."

Table 1. — Comparison of nearshore and offshore morphometric characteristics of Lake Michigan, excluding Green Bay.

	Nearshore	Offshore	Lake
Depth range (m)	0-30	30-275	0-275
Water surface (km^2)	11300	45200	56500
Water surface (%)	20	80	
Volume (km^3)	220	4580	4800
Volume (%)	4.6	95.5	
Mean depth (m)	19.1	101	85

The need to examine the dynamics of large lakes from a different perspective than that for smaller lakes is to a great extent a function of physical factors of these large systems. Lake Michigan has a long residence time: the volume divided by the outflow is approximately 100 years. It has a mixing time of about 180 days (Boyce, 1974) which is long relative to time scales for phytoplankton growth. Hypsographic relationships show that 20 percent of the area but only 4 or 5 percent of the volume is contained within the 30-meter contour (Table 1). Finally, because of the large size and thus, great variations in depth, the water surface warms differentially. The shallower nearshore waters warm more rapidly in the spring causing a thermal bar to form that separates the nearshore from the offshore (Mortimer, 1975). The thermal bar produces sharp nutrient gradients and has a pronounced effect on the distribution and abundance of phytoplankton (Stoermer, et al. 1968; Davis, et al. In press).

In this paper I will show that the nearshore zone receives much greater phosphorous loads, produces greater standing crops of algae, and contains different species composition of phytoplankton than the offshore waters. The nearshore phytoplankton differences can be attributed to the combined effects of anthropogenic materials, including phosphorus. In total these results are significant for lake protection because the major uses of water are in the nearshore, whereas most research is directed at offshore problems.

PHOSPHORUS LOADING

The four major sources of phosphorus loading are tributary flows, atmospheric inputs, direct municipal discharges, and shoreline erosion (Table 2). Estimated total and source inputs by different investigators vary considerably; however, given the size of this particular lake and its drainage basin and the limited study on inputs these variations are not unexpected. All estimates agree in that the tributary loading is the greatest. Large discrepancies exist for estimates of shoreline erosion, ranging from 1.35 to 3.7×10^6 kg/yr, and for atmospheric inputs, ranging from 1.0 to 1.69×10^6 kg/yr. The main reason for the larger number of 1.69 10^6 kg/yr (Eisenreich, et al. 1977) for atmospheric inputs is that dry fallout was not included in the estimated input of 1.0×10^6 kg/yr (Murphy and Doskey, 1976). Absolute values for different sources are not critical for the points emphasized in this paper.

In addition to uncertainties about the magnitude of the loads there is, of course, the well-recognized problem of the proportion of any given load that is available to phytoplankton for growth. No attempt will be made in this paper to address this because unequal load distribution can be shown without addressing the question of availability.

Disproportionate loading of phosphorus from tributaries to different shoreline zones has been recognized for some time. Schlelske (1975) pointed out that as much as 40 percent of the tributary phosphorus loading to Lake Michigan could be attributed to inputs of the Grand, St. Joseph and Kalamazoo rivers. All are located within 120 kilometers of shoreline on the southeastern part of the lake. More recently Sonzogni, et al. (1978) reported that the same tributaries in 1975 and 1976 supplied 48 and 46 percent of the total tributary phosphorus loading to Lake Michigan including Green Bay (Figure 1). Roughly half of the tributary phosphorus loading therefore was concentrated within a 120-kilometer length of shoreline; nearly 25 percent of the total came from one tributary, the Grand River.

Table 2. — Sources of total phosphorus input to Lake Michigan for three time periods. All inputs are 10^6 kg/yr.

	1974[1]	1975[1]	1975-76[2]
Direct industrial discharge	0.05	0.06	
Direct municipal discharge	1.09	1.07	
Tributary inputs	4.97	4.23	3.39
Municipal point			(1.04)
Industrial point			(0.22)
Shoreline erosion	1.35	3.7	
Atmospheric inputs	1.00	1.69	
	8.46	10.75	

[1] Eisenreich, et al. 1977.

[2] averages of data for 1975 and 1976 are from Sonzogni, et al. 1978.

Loadings from direct municipal discharges to the lake are also disproportionate. This is readily illustrated by some reports that as much as half of the total phosphorus loading from direct municipal discharges originates from the City of Milwaukee. That direct municipal discharge of approximately 500 metric tons is roughly half the municipal discharges to tributaries of 1,191 metric tons/yr (Sonzogni, et al. 1978). It is also a phosphorus load equivalent to that from the Fox River and larger than any tributary load other than the Grand River.

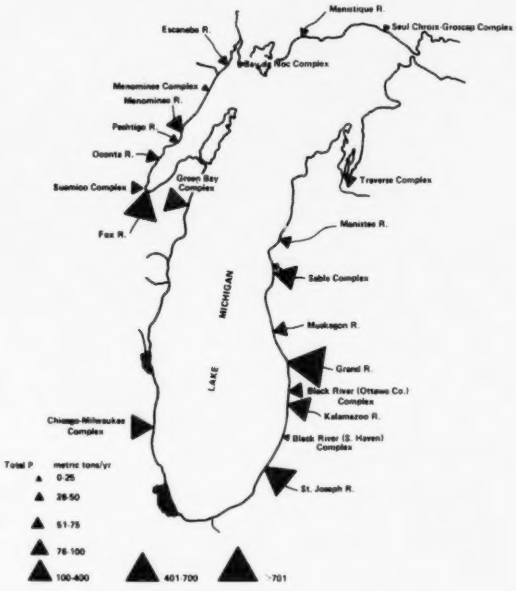

Figure 1. — Magnitude of total phosphorus tributary loadings to Lake Michigan. Data in metric tons/yr from Sonzogni et al. 1978.

Atmospheric loadings also are not uniform over the lake basin, although the variation is not as great as that for tributary loading or direct municipal discharges. According to Eisenreich, et al. (1977) atmospheric loading to the southern basin is roughly twice as large as to the northern basin. Although atmospheric loadings comprise a relatively large part of the total phosphorus loading to the lake, as much as 20 percent by some estimates (Table 2), the relative effect in the lake probably differs from that of either municipal or tributary inputs. Atmospheric inputs are distributed over the entire lake surface whereas loadings from tributaries and municipal sources are primarily to the nearshore zone.

Nearshore phosphorus inputs are distinguished functionally from atmospheric inputs in that phosphorus loaded to the nearshore zone must be transported through the nearshore waters prior to being mixed with the offshore waters of the lake. Phosphorus transported through the nearshore zone is acted on by biological processes before it is diluted with the open lake waters, whereas most of the atmospheric input is transported directly to the open lake.

Using data on phosphorus loading (Table 2) and morphometric data (Table 1) I have calculated areal and volumetric phosphorus loads to the nearshore and offshore zones. These data clearly show that loading in the nearshore zone is disproportionate to that in the offshore waters. Tributary loadings are 20 times greater in the nearshore than in the offshore zone on a volumetric basis and 5 times greater on an areal basis (Figure 2).

It should be obvious that these average loads do not reflect the absolute range in loadings that occur within the lake. For example, because 50 percent of the tributary loadings result from the three tributaries in the southeastern part of the lake these loads represent greater than average nearshore loading (Figure 1). Likewise, some nearshore areas in the northern part of the lake receive relatively small phosphorus loads, resulting in smaller than average loading.

In summary, of the three areas where phosphorus loading to the nearshore zone is greater than the average, only two are in the main body of Lake Michigan: (1) The 120-kilometer length of shoreline in the south where the Grand, St. Joseph, and Kalamazoo rivers drain to the southeastern part of the lake; (2) the area on the west shore directly across the lake from the Grand River which receive municipal input from the city of Milwaukee; (3) the southern end of Green Bay where the Fox River is the major source of input. Nutrient effects on biological processes in this area are most evident in the southern part of Green Bay and become less evident with distance north from the mouth of Fox River.

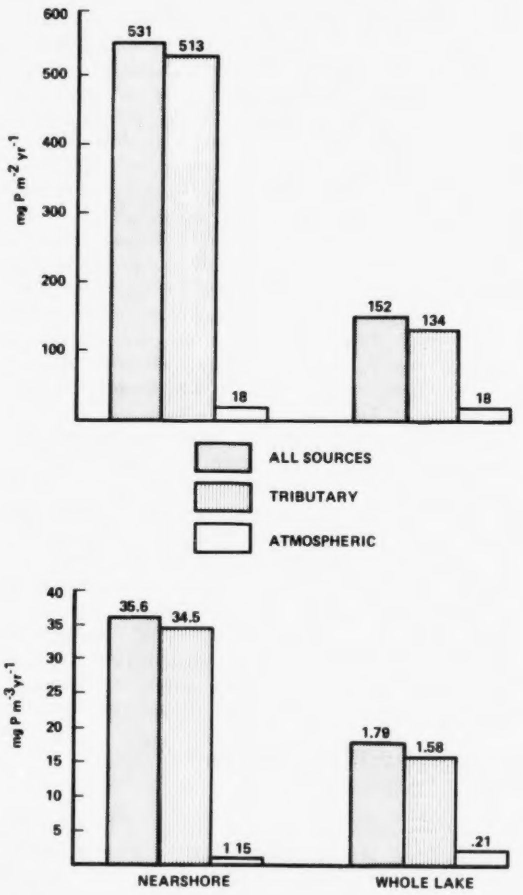

Figure 2. — Areal (upper) and volumetric (lower) total phosphorus loading to Lake Michigan. Loads are calculated for the nearshore zone and whole lake. Nearshore is set at 25 percent of the surface area. See Table 1 for other morphometric data.

BIOLOGICAL CONSEQUENCES OF NEARSHORE LOADING

Given the disproportionate loadings to the nearshore zone, the biological manifestations of nutrient enrichment are most pronounced in this area of the lake. Chlorophyll standing crops are several times larger on the average in the nearshore zone than in the offshore zone (Table 3), both along the southeastern shoreline where tributary inputs are the greatest, and also on the southwestern shoreline where tributary inputs are not as large a factor (Figure 3).

Not only is there a difference in standing crop but there is also a pronounced difference in species composition between the nearshore zone and the offshore zone. A number of species characteristic of highly enriched or polluted waters have been found in enriched nearshore zones (Stoermer and Yang, 1970). It has been found that species of *Melosira* tolerant of nutrient enrichment dominated the nearshore zone and were replaced by species less tolerant of enrichment in offshore waters (Holland, 1968).

Recently we have completed studies on the distribution of nutrients and their relationships to species composition and standing crop of phytoplankton in the nearshore zone (Schelske, et al. 1980). These studies showed that, as expected, standing crops of phytoplankton were greater and species composition different in nearshore waters than in offshore waters. In addition, we showed that the nutrient input from rivers influenced the biological characteristics over a considerable distance offshore from the mouth of the tributary. In areas affected by tributary inputs phytoplankton composition was dominated by species supplied with the tributary inflow and varied from river to river and with the season of the year. Inshore-offshore differences were less pronounced in the northern part of the lake where tributary phosphorus loading (Figure 4) was relatively small compared to the southern basin.

Phytoplankton species composition in tributary inputs obviously differed from that in nearshore waters so that the major dominants in the tributaries could be used as biological tracers for river inputs (Schelske, et al. 1980). The plume of the Grand River with its characteristic phytoplankton, for example, extended 1.6 kilometers offshore, but beyond this point the nearshore phytoplankton were characteristic of enriched areas such as the transects offshore from Milwaukee and the Kalamazoo River where there was no large tributary input. Based on phytoplankton species composition and chlorophyll standing crops, the nearshore zone extended offshore at least 6.4 kilometers and on some transects 13 kilometers offshore.

Table 3. — Comparison of chlorophyll and phosphorus concentrations in nearshore and offshore waters of Lake Michigan. Values for the lake are based on volume-weighted averages for the nearshore and offshore waters. See Table 1 for volumetric data.

	Nearshore	Offshore	Lake
Average total phosphorus[1] (μg P/liter)	15.2	8.1	8.4
Average chlorophyll[1] (μg/liter)	4.7	2.2	2.3
Maximum spring chlorophyll[1] (μg/liter)	14.0	4.4	--
Average spring chlorophyll[2] (μg/liter)	12.0	2.3	--

[1] Rousar (1973). Nearshore, 4.8 km from Milwaukee.
[2] Ladewski and Stoermer (1971). See Fig. 3.

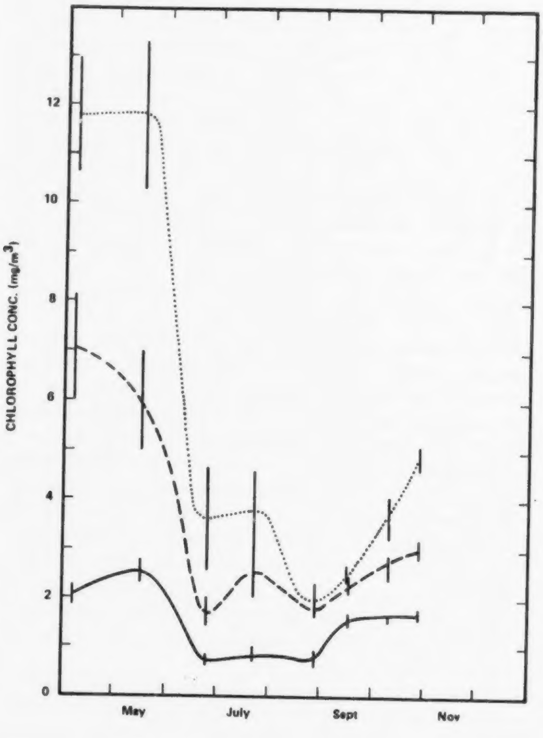

Figure 3. — Chlorophyll concentration in 1971 averaged by depth range and month. Key: Dotted line shows mean value for stations between 10 m and 40 m deep and solid line shows mean value for stations deeper than 40 m. For each cruise there are nominally 12 stations shallower than 10 m, 16 between 10 and 40 m deep and 13 deeper than 40 m. Error flags show the standard error of the mean. (Ladewski and Stoermer, 1973).

Greater standing crops of phytoplankton in the nearshore zone on first analysis appear to be directly attributable to tributary nutrient loading, particularly phosphorus. Effects of enrichment, however, appear to ring at least the southern basin of Lake Michigan, and therefore the effect may not be due only to tributary loading because it has already been shown that tributary nutrient loading on the western shore is less than that for the eastern shore of the southern basin (Figure 1). It has also been reported that chlorophyll standing crops in the nearshore zone off Milwaukee were several times greater than those in the offshore waters, a difference attributed to municipal phosphorus discharges at Milwaukee (Rousar, 1973). Further analysis of inshore-offshore differences are caused not only by greater loading of nutrients to the nearshore zone but also by physical factors and biological and chemical processes that are not clearly understood (Beeton and Edmondson, 1972).

One important factor in considering phosphorus loadings to the nearshore is the time response of phytoplankton relative to such loadings. Time responses for phytoplankton growth vary depending on the physiological state of phytoplankon. If phytoplankton respond immediately without a lag phase, effects of added nutrients might be expected to occur within a 4 or 5-day period. If it is assumed that phytoplankton cells divide at a rate of about one per day or slightly less, then within this 4 or 5-day period standing crops could increase by a factor of 10. If the response lag to enrichment were 1 to 3 days this time would be extended to a 4 to 8-day period. Given average coastal currents in the nearshore environment of .5 km/hr one would then expect the phosphorus to be transported no more than 50 to 100 kilometers from the source before it was used by phytoplankton in the coastal zone. Since current reversals are frequent in the nearshore zone one would not expect the affected area to extend as far as 50 to 100 kilometers from the source very frequently.

The preceding calculations consider only effects of phosphorus on growth of phytoplankton in the nearshore zone and neglect any recycling of phosphorus or transport of phosphorus in phytoplankton to other areas where it can be recycled and used again for phytoplankton growth. Considering the long time constants for physical transport and mixing of phosphorus throughout the lake basin relative to the time constants for uptake and growth by phytoplankton, phosphorus is either recycled many times through the plankton community or carried within the lake by mechanisms other than simple mixing and diffusion.

NUTRIENT ENRICHMENT EXPERIMENTS

Experimental work with the effects of nutrients on growth and species composition of natural phytoplankton can provide insight into why species composition differs between the nearshore and the offshore zones. These experiments show that growth rates of offshore natural phytoplankton assemblages can be increased by adding phosphorus alone and that greater growth rates can be obtained if vitamins, trace metals, and a chelating agent are combined with phosphorus additions (Figure 4). This effect has been substantiated in experiments with water from Lake Michigan (Schelske, et al. 1974), Lake Superior (Schelske, et al. 1972), and Lake Huron (Lin and Schelske, 1979). The specific agent that caused greater phytoplankton responses was identified in one factorial experiment. Analysis of variance showed that the chelating agent, EDTA, produced a statistically significant main effect and that vitamin and trace metal additions did not (Schelske, et al. 1978). Higher growth rates with additions of vitamins, trace metals, and chelating agents indicate that the system may be "nutrient saturated" with this treatment. Nutrient saturated will be used in the paper to describe the experimental conditions under which maximum growth rates were obtained even though this condition has not been verified experimentally.

In our experiments phytoplankton are sensitive to phosphorus enrichment and are phosphorus saturated at a relatively low level, at concentrations ranging from 5 to 15 μg P/liter. The absolute concentration is not important because this would undoubtedly change depending on experimental conditions. However, it is important to note that with the addition of trace constituents and phosphorus, growth rate is no longer limited at low phosphorus levels and increases with increasing phosphorus concentrations (Figure 4).

The presence of trace constituents and phosphorus increases not only the growth rate but also the yield or standing crop (Figure 4). Because yield is ultimately affected by availability and quantity of nutrients it seems obvious that a greater yield would result from nutrient-saturated growth than from nutrient-limited growth.

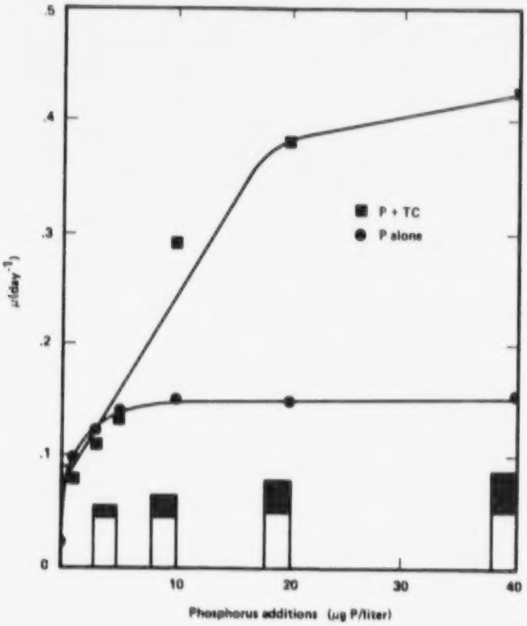

Figure 4. — Relationship between phosphorus additions with and without trace constituents (TC) and growth rate in doublings/day. Bar graphs indicate final yield for enrichments of 5, 10, 20 and 40 μg P, solid portion represents increases due to trace constituents.

Perhaps the most important result of these experiments is the demonstration that trace constituents added with phosphorus change the species composition in the phytoplankton assemblage. Stoermer, et al. (1978) showed that the phytoplankton species succession resulting from phosphorus additions alone closely paralleled that which occurred in Grand Traverse Bay from which the water had been obtained originally. *Fragilaria crotonensis* was the major dominant in Grand Traverse Bay during and after the experiment. This species was also the major dominant in laboratory experiments which received only phosphorus enrichments. Under nutrient-saturated growth conditions resulting from trace constituent enrichment, species succession was different. The major dominant shifted to *Stephanodiscus subtilis* (Stoermer and Yang, 1970) along with other nutrient tolerant species of *Stephanodiscus*, including *S. tenuis* and *S. minuteus* (Schelske, et al 1980).

DISCUSSION

Data show that from the standpoint of eutrophication processes large lakes the size of Lake Michigan should be considered as two separate systems, a nearshore and an offshore area. The offshore volume is much larger than the nearshore so average conditions in the lake are mainly determined by offshore properties even though conditions are greatly different in the nearshore, (Table 3). These data indicate that empirical models of the Vollenweider type are adequate to address the relationship between average standing crop and chlorophyll concentrations for the open lake water mass. Such models, however, were not designed and should not be used to evaluate water quality in the nearshore zone where conditions in time and space are highly variable.

Disproportionate loading of phosphorus (Figure 2) is obviously one of the factors contributing to greater standing crops of phytoplankton in the nearshore (Table 3). Phytoplankton assemblages characteristic of enriched nearshore zones apparently cannot be attributed only to phosphorus enrichment because data have been obtained that show succession of species in assemblages is affected by trace materials (EDTA, vitamins, and trace metals) associated with algal nutrition (Stoermer, et al. 1978; Lin and Schelske, 1979). In addition, the presence of these minor constituents stimulate growth rates to greater levels than those realized from phosphorus additions alone (Figure 4). Anthropogenic inputs to the nearshore would be expected to enrich this area with vitamins, trace metals, and chemical compounds with chelating capacities.

In large lakes, symptoms of nutrient enrichment or eutrophication are first evident in the nearshore and later may be present in the entire lake basin as was the case with Lake Erie (Beeton and Edmondson, 1972). In Lake Michigan enrichment effects presently are most evident in the nearshore. Whether this ring around the lake will or could eventually cover the lake surface as it did in Lake Erie is open to conjecture. Greater nutrient loads would be required in Lake Michigan to cause an effect comparable to Lake Erie because nearshore waters are a relatively smaller proportion of the total water mass and Lake Michigan is deeper.

Managing phosphorus inputs to Lake Michigan is still the most critical problem in controlling eutrophication. To date, little attention has been given to critical nearshore effects. Instead, considerable effort has been devoted to calculating total phosphorus budgets which are then used in lake basin models. This approach is necessary for determining phosphorus contributions to downstream lakes and to evaluating long-term trends in open lake water quality, but it is not directly applicable to the problem of nearshore water quality. Any efforts to model nearshore water quality will be hampered by the lack of a long-term data base from which the model could be verified.

To be useful a model of nearshore water quality would have to incorporate several features of the model that was developed for Saginaw Bay, including limitation of phosphorus, silica, and nitrogen for phytoplankton growth and capability to adjust phytoplankton forms to nutrient conditions (Bierman, et al. In press). Although it is possible to model succession of phytoplankton forms, i.e., from diatoms to non-diatoms as the result of silica limitation or to nitrogen-fixing blue-greens as the result of nitrogen depletion, there are at present no experimental data on which to base the more complex modeling that would be required for models of succession in multi-species assemblages.

Natural phytoplankton assemblages should be used to determine effects of nutrient enrichment in oligotrophic or mesotrophic waters because the quality of phytoplankton in many cases is as important as the quantity. Effects of perturbations of any type on species succession and dominance can be studied only with natural assemblages or possibly with several cultured species. Data presented in this paper indicate that μ_{max} and other kinetic parameters may vary with the trace constituents in natural water or with trace nutrient conditions in artificial media. That trace constituents (EDTA, vitamins, and trace metals) increased growth rates markedly (Figure 4) points to the problem of simulating natural conditions in artificial media. These considerations also point to the need to determine a true μ_{max} (maximum growth rate) that would occur under nutrient saturated conditions in experiments. Possibly this growth rate should be based on calculated quantum photosynthetic yields which would represent a true maximum and provide a basis on which other rates could be compared. Because physical, chemical, and biological components of natural systems are dynamic, experiments to determine the responses of natural phytoplankton assemblages must be conducted frequently so the influence of these changing conditions on biological processes can be evaluated (Lin and Schelske, 1979).

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MODELING THE RESPONSE OF THE NUISANCE ALGA, *CLADOPHORA GLOMERATA*, TO REDUCTIONS IN PHOSPHORUS LOADING

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ABSTRACT

A mathematical model was developed to evaluate the impact of various phosphorus management strategies on nuisance growths of the filamentous alga *Cladophora glomerata*. The model was supported by intensive ecological studies and an extensive field monitoring program. The results of simulating spatial and seasonal variation in algal biomass and associated nutrient parameters agree well with field observations. The calibrated model is used to predict the response of the system under study to a demonstration phosphorus removal program. Implications to large-scale phosphorus management strategies are discussed.

INTRODUCTION

Cladophora glomerata, an attached filamentous green alga (Chlorophyceae), is a recognized nuisance in the littoral region of the Laurentian Great Lakes and many smaller inland lakes. Nuisance growths of this organism in small inland lakes are typified by those observed in the Madison (Wisconsin) lakes, particularly Lake Mendota. In the Great Lakes, water quality is most severely impacted by this alga in Lakes Erie and Ontario (Shear and Konasewich, 1975). In these lower Great Lakes massive accumulations of rotting algal material has resulted in closed beaches, decreased lakeshore property values, and reduced utility of the environment as a recreational resource. The proliferation of *C. glomerata* in Lakes Erie and Ontario is thought to be related to lakewide nutrient enrichment rather than simply point discharges of nutrients. Site specific occurrences of the alga have been reported from Lakes Huron, Michigan, and Superior (Niel and Owen, 1964; Lin, 1977; and Herbst, 1969). The offshore waters of these lakes cannot support significant growth of *Cladophora* because of low phosphorus levels.

The presence of abundant growths of *C. glomerata* may indicate an overall or local reduction in water quality. In small inland lakes septic tank drainage may encourage local growth of plant material or may elevate lakewide nutrient levels so that plant growth is prolific throughout the littoral region. *C. glomerata* is naturally present in rivers, streams, and many inland lakes. Its increase to nuisance proportions, however, reflects a serious perturbation of the quality of our inland waters. As such, amelioration of nuisance conditions by reducing phosphorus loading rates would reflect well upon our commitment toward the lessening of man's impact on the environment.

In 1978, the University of Michigan, in cooperation with the EPA Large Lakes Research Station at Grosse Ile, Mich., began a 3-year program to examine the potential for reducing nuisance growths of *Cladophora* in the Great Lakes. The key function of the project is to develop a mathematical model relating the production of *Cladophora* biomass to phosphorus loadings to the Great Lakes. Such a model would be useful in evaluating the impact of various phosphorus management strategies in controlling *Cladophora* growth. The model integrates available information on the alga as well as indicating areas where new basic studies on the ecology of the organism are warranted. These topics serve as subjects for special investigations designed directly to support the model. A field monitoring program at a site known for nuisance *Cladophora* growth provides data for calibration of the model. Observations following a demonstration phosphorus removal program at the site verify the utility of the model. Projections regarding the impact of various phosphorus management strategies at this and other sites may be examined by using the proper set of loading rates and boundary conditions associated with that location.

FIELD SITE

A field site with a known *Cladophora* problem was selected from which to gather data on the growth of the alga for use in calibrating the mathematical model. A site was chosen which was perturbed by a single major nutrient source, isolated from the complexities of whole-lake growth forcing conditions, e.g., Lake Erie. A site was selected at Harbor Beach, Mich. on Lake Huron. The water quality of Lake Huron is such that significant growth of *C. glomerata* cannot be supported by offshore waters.

Harbor Beach has a population of approximately 2,000 and is an agricultural and light industrial community. The Harbor Beach wastewater treatment plant is a high-rate trickling filter operation with a design flow of 350,000 gallons/day. The soluble reactive phosphorus loading from the plant for 1979 was 2.163 metric tons (5.93 kg/day). The plant discharges to Spring Creek, a small stream with good upstream water quality. The wastewater treatment plant phosphorus loading results in an annual mean soluble reactive phosphorus concentration of approximately 700 μgP/l at the mouth of Spring Creek. Nuisance growths of *C. glomerata* occur in a symmetrical pattern about the mouth of Spring Creek. the region of perturbation extends 2 kilometers south of the discharge and 0.5 kilometers north where the area suitable for growth is truncated by the presence of a dredged manmade harbor. The presence of extensive cobble shoals (substrate required for attachment) and quiescent bays (reduced mixing with offshore waters) contribute to the potential for algal problems.

The isolation of the site from whole-lake or multiple source nutrient perturbations is confirmed by chemical and biological data. Dissolved phosphorus levels and that stored in algal cells (internal phosphorus) decline with increasing distance from the loading point, eventually reaching background or boundary levels. *Cladophora* biomass decreases as well, with no observable growth at the station most removed from the nutrient source (1.8 km). The relationships between chemical and biological parameters and distance from the nutrient source have been described in detail in an earlier publication (Auer and Canale, 1980).

MODELING FORMAT

The mathematical model used in this project was developed to fully use support available from the specialty studies and monitoring program. The model is composed of a fluid transport and a kinetic submodel. The function of the former is to relate nutrient loading and advective and dispersive transport so that the distribution of dissolved phosphorus in the study area may be predicted. The resultant soluble reactive phosphorus concentrations become the forcing parameter for nutrient uptake rates, a component of the kinetic submodel. Equation 1 summarizes the primary factors regulating the growth of *Cladophora* at the study site on Lake Huron. These factors are the components of the kinetic submodel.

$$\mu = \hat{\mu}\,(f_1(I) \cdot f_2(T) \cdot f_3(Q) \cdot f_4(X)) - R - S \qquad \text{Eq. 1}$$

where: μ : specific growth rate
$\hat{\mu}$: maximum specific growth rate
$f_1(I)$: function relating growth to light intensity
$f_2(T)$: function relating growth to temperature
$f_3(Q)$: function relating growth to internal phosphorus
$f_4(X)$: function relating growth to carrying capacity
R : respiration rate
S : sloughing loss

TRANSPORT SUBMODEL

Spatial resolution for the model is achieved through establishing completely mixed cells with flows reflecting wind-driven and wave-induced current regimes. The model cells are oriented in two layers parallel to the shoreline. Current regimes are calculated with classical momentum equations which include the effects on non-linearities, wind, bottom friction, and wave action. Several intensive chloride grids as well as weekly and daily chloride measurements at selected stations were also combined with daily wind observations to gain an understanding of nearshore current regimes.

Nutrient loading from the wastewater treatment plant was monitored twice weekly. Soluble reactive phosphorus and total phosphorus concentrations were measured weekly at 25 nearshore stations representing offshore and longshore boundary conditions. Sampling was conducted from ice-out through November, with loading measurements continuing through the winter. The transport submodel then considers loading data and current regimes as well as algal uptake in establishing soluble reactive phosphorus concentrations throughout the study area. These values may be compared with weekly monitoring data from lake stations to calibrate the model.

KINETICS SUBMODEL

The kinetics submodel considers each factor thought to contribute importantly to the growth of *Cladophora glomerata* at the study site on Lake Huron. Three growth terms are considered: Light, temperature, internal phosphorus level, and carrying capacity. The latter is dynamically related to dissolved phosphorus levels through nutrient uptake kinetics. Loss terms include respiration and sloughing. Sloughing refers to the separation of algal filaments from the substrate, leading to shoreline deposition of algal material. The model calculates the specific growth rate and ultimately the algal biomass throughout the study site by relating these factors in the fashion described previously in Equation 1. It is useful to examine the derivation and data base associated with the major components of the kinetic submodel.

LIGHT

An experiment conducted at the BIOTRON facility at the University of Wisconsin examined the relationship between growth rate and light (and temperature). Isolates of *C. glomerata* obtained from Lake Huron and two small inland lakes were cultured over a matrix of light and temperature levels in a crossed-gradients room. Carbon uptake was measured over a range of light intensities (60 to 1,000 μE/m^2 sec) and the resultant specific growth rate was calculated. These measurements correspond to net photosynthesis. Gross photosynthesis was calculated by adding to the net photosynthesis data a factor representing the respiration rate as derived from the literature (Jackson, 1966). The results of this experiment are presented in Figure 1. These data show growth to be a hyperbolic function of light intensity, with saturation at high light levels. Independent measurements were made to determine the maximum specific growth rate. The data of Figure 1 were then modified to reflect this information. The range of light intensities used in this experiment are representative of those observed in the field. All field and laboratory measurements recorded photosynthetically-active radiation (PAR) using a quantum meter.

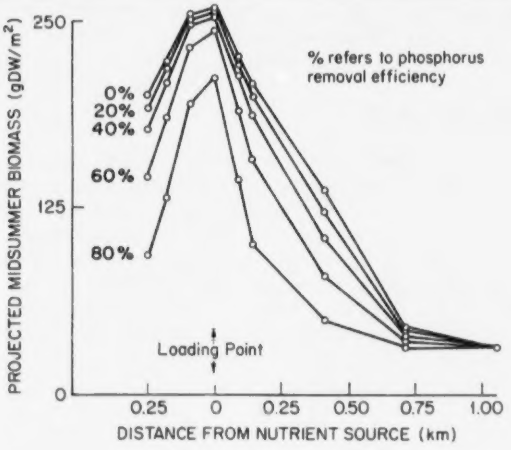

Figure 1. — The relationship between specific growth rate and light intensity for *C. glomerata*.

The light function may be used to calculate daily specific growth rates at depth if input data in the form of light at depth are provided. The level of light at specific depths was calculated on a daily basis. A series of measurements was made to establish a relationship between the extinction coefficient for light and Secchi disk transparency. Depths across the study area were obtained through extensive mapping and daily water level measurements. Daily estimates of incident solar radiation were obtained from the literature and corroborated on site. These estimates were used with the extinction coefficients resulting from daily Secchi disk readings to calculate light at depth. This value served as input to the hyperbolic light function, through which daily values for growth rate as a function of light intensity could be calculated. The results of these calculations indicated that for this site on Lake Huron, light limits the growth of *C. glomerata* below a depth of approximately 1.25 meters.

TEMPERATURE

Temperature has been considered an important factor in regulating growth of this nuisance alga. Data from the BIOTRON experiments were used to describe the function relating growth rate to temperature. Again, carbon uptake rates were measured at 2, 5, 10, 15, 20, 25, 30, and 35°C. A value for gross photosynthesis was calculated by adding the curve for respiration as a function of temperature to the curve for net photosynthesis. The results of this experiment are presented in Figure 2. The growth rate was observed to increase in an approximately linear fashion with temperature to an optimum range of 20 to 30°C. Severe inhibition was noted above 30°C. The importance of temperature in the nearshore Lake Huron environment is seen most clearly in the spring and fall cold periods. Temperature inhibition is not observed as water temperatures seldom exceed 23°C. Water temperatures in the May to August growing season are generally within the optimum temperature range of the alga.

The actual calculations of growth rate as a function of temperature are made possible through the availability of daily temperature data at the study site. Temperature also affects respiration. This relationship is discussed in a later section.

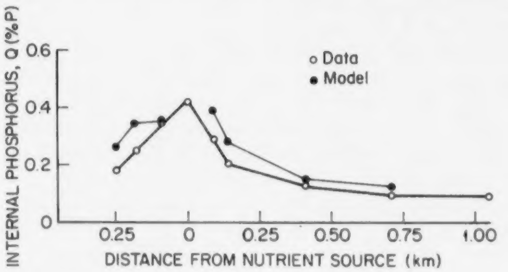

Figure 2. — The relationship between relative growth rate and temperature for *C. glomerata*.

INTERNAL PHOSPHORUS POOLS

Droop (1973) and Senft (1978) have discussed the importance of internal nutrient pools in regulating the growth rates of algae. Experiments examined the relationship between internal phosphorus pools and the growth rate of *C. glomerata*. Algal material was harvested from sites of varying proximity to the nutrient source; these samples thus provided a range of internal phosphorus levels. Internal phosphorus levels (Q) are measured by the persulfate digestion techniques and expressed as the percent P of dry weight (%P). Rates of photosynthesis and respiration were measured under constant conditions of light and temperature, and the resulting specific growth rates were calculated. The results, illustrated in Figure 3, show that growth rate increases with increasing pool size, saturating at high internal phosphorus levels. It is important to note that growth rate increases rapidly as the internal pool size rises above the minimum cell quota (Q_0). Very little additional growth is gained for considerable increases in pool size as the saturation level is approached. This phenomenon will have serious implications for phosphorus management strategies. The position of the algal community along the response curve will determine to a large extent the success of any phosphorus removal efforts.

The mathematical model calculates internal phosphorus pools directly given a growth rate (loss term for internal pools) and the kinetic of nutrient uptake (gain term for internal pools). Experimentally derived relationships for phosphorus uptake kinetics are used in the model. Phosphorus uptake is regulated by two factors: Dissolved (external) phosphorus concentrations and internal phosphorus concentrations. Phosphorus uptake rates have been demonstrated experimentally to increase with increasing external phosphorus levels in the classic Michaelis-Menten fashion. Saturation is reached at very high substrate levels relative to that observed for phytoplankton. Increases in internal phosphorus concentrations reduce phosphorus uptake rates through negative feedback. This process is a form of enzyme inhibition and has been discussed by Rhee (1978). A kinetic structure relating the substrate saturation process and feedback inhibition has been derived experimentally for *C. glomerata* and included in the model. These experiments will be described in detail in a later publication.

The model may be calibrated by comparing output with measured values for internal phosphorus levels obtained from stations throughout the study area. Internal phosphorus levels are measured at 28 stations at weekly intervals throughout the growing season. These data confirm the longshore symmetry of internal phosphorus concentration and define phosphorus as the element limiting growth in the shallow regions of the nearshore. Very little variation in the level of internal phosphorus was noted for distance offshore at a given station. This indicates that light is the important factor controlling growth in the offshore direction.

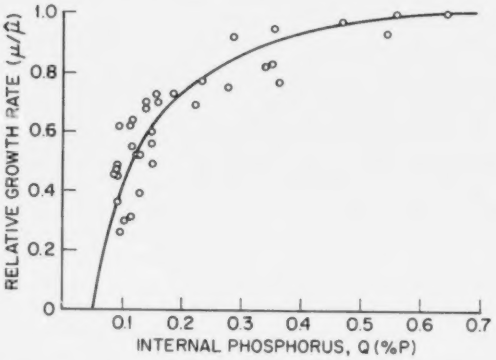

Figure 3. — The relationship between relative specific growth rate and internal phosphorus concentration for *C. glomerata*.

CARRYING CAPACITY

A carrying capacity term has been included in the model to reflect spatial limitations on the substrate as well as self-shading effects. An empirically derived value for the maximum attainable biomass (X) of 600 gDW/m² is employed. As calculated biomass approaches this level, the model term reduces the growth rate through negative feedback.

RESPIRATION

Respiration represents the most important continuous loss term for the mathematical model of *Cladophora* growth. Experiments currently in progress at the BIOTRON facility will carefully define the relationship between respiration, temperature, and overall metabolic activity. At the present time Jackson's data (1966) are used to derive the relationship between respiration and gross and net photosynthesis. A linear relation between respiration and temperature is used in the model. This function is described in Equation 2. Calculated values for respiration rate are input to the model to obtain the net specific growth rate as a function of temperature.

$$R = R^{*}\,(T/20) \qquad \text{Eq. 2}$$

where: R : respiration at T°C
R* : respiration at 20°C
T : measured temperature

SLOUGHING

In that *Cladophora* is not grazed to any extent, the only other loss term is sloughing. Sloughing occurs when severe mechanical disruption causes the algal filaments to become unattached from their substrate and float free in the water column. Shoreline deposition and nuisance accumulation generally result from this process. Although overall physiological condition is thought to bear importantly on sloughing, we have been quite successful in relating sloughing events directly and solely to severe storm (high wind) events. For the current model, sloughing is related empirically to the occurrence of storm events. Experiments in progress with *in situ* algal populations will better relate those storm events, standing crop, and magnitude of sloughing loss.

BIOMASS

The end product of the model is the prediction of standing crop of *Cladophora* biomass. Biomass is accumulated in the model as the product of the growth rate and the current standing crop. The standing crop of *Cladophora* biomass at 14 stations at the Lake Huron study site is measured weekly by harvesting representative samples of the alga and substrate. Density of coverage and areal distribution are measured as well. The results are expressed as grams of oven-dried algal material per square meter of substrate (gDW/m²). The most important calibration of the model involves comparison of model generated biomass data with that observed through the growing season.

RESULTS OF MODEL CALIBRATION

An understanding of the growth dynamics of *Cladophora* entails both spatial and temporal resolution. Generally, temporal or seasonal dynamics are more difficult to simulate. Figures 4 through 6 compare model output of spatial variation for soluble reactive phosphorus, internal phosphorus, and biomass with observed annual average values. In the case of biomass, the midsummer mean value is used to better reflect the maximum standing crop. Model agreement for the two phosphorus components is quite good, accurately reflecting the reduction in dissolved and stored phosphorus with increasing distance from the loading point. Biomass simulation is also quite good, especially considering the heterogeneity of the nearshore substrate (mixed sand, gravel, and cobbles).

Figures 7 and 8 compare model output with monitoring data describing the seasonal variation in biomass and internal phosphorus for stations near the nutrient source. The mean for each sample date for six stations is plotted. Internal phosphorus levels are relatively constant throughout the year for these stations because of continuous loading from the wastewater treatment plant. The biomass data illustrate temperature limitation in the spring and fall as well as the rise to a maximum standing crop in June through August. Superimposed upon the smooth curve for maximum standing crop is the impact of sloughing loss. The sharp declines in biomass are major sloughing periods and have been empirically associated with storm events. In most cases, biomass levels return rapidly to their pre-slough levels as the restrictions of carrying capacity and substrate/light competition are relaxed following sloughing. This excellent match between model calculated values for biomass and the phosphorus components and observed data allow projections to be made regarding the impact of various levels of phosphorus removal on nuisance growth of *Cladophora* at the site.

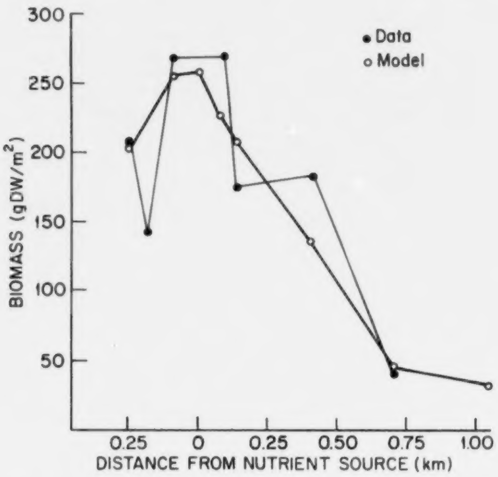

Figure 4. — Comparison of model output and observed data for the distribution of soluble reactive phosphorus about the nutrient source. Values are annual means.

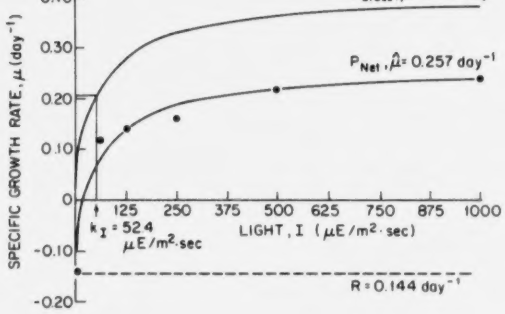

Figure 5. — Comparison of model output and observed data for the distribution of internal phosphorus levels about the nutrient source. Values are annual means.

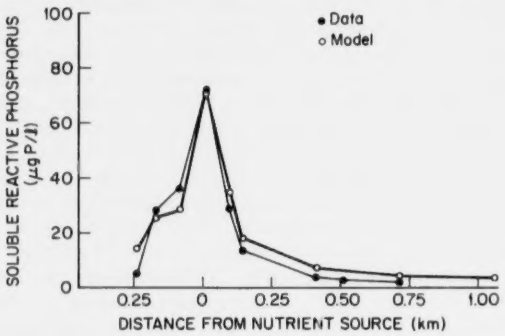

Figure 6. — Comparison of model output and observed data for the distribution of *Cladophora* biomass about the nutrient source. Values are midsummer means.

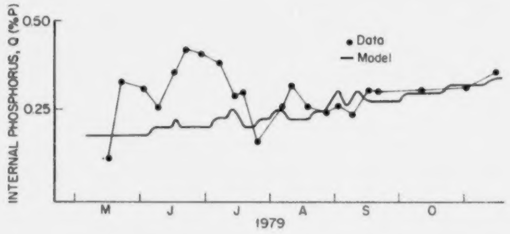

Figure 7. — Comparison of model output and observed values for the seasonal variation in *Cladophora* biomass. Data are mean values for six stations near the nutrient source.

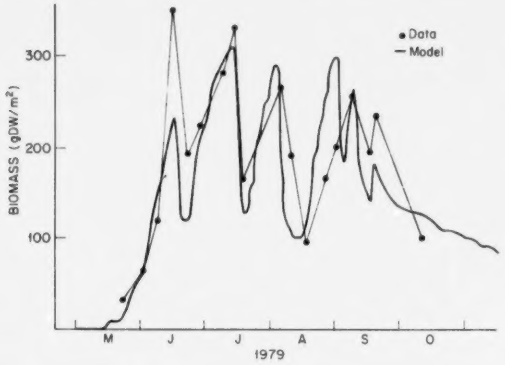

Figure 8. — Comparison of model output and observed values for the seasonal variation in internal phosphorus levels. Data are mean values for six stations near the nutrient source.

DEMONSTRATION PROGRAM AND MODEL PROJECTIONS

In February of 1980 a demonstration phosphorus removal program was instituted at the Harbor Beach wastewater treatment plant. Phosphorus was removed by alum precipitation with a polyelectrolyte coagulant aid. Initial results indicate that an 80 percent reduction in soluble reactive phosphorus at the mouth of Spring Creek may be anticipated. The calibrated model is used to project the impact of various levels of phosphorus reduction on the standing crop of *Cladophora* at the site. In this manner, the accuracy of the calibrated model may be verified. Figure 9 describes the spatial distribution of midsummer biomass levels associated with several degrees of phosphorus removal. The impact is most noticeable at points remote from the nutrient source. Such a result is consistent with the relationships presented in Figures 3 and 5. Algal material distant from the nutrient source has lower levels of internal phosphorus (0.10 percent) and lies, therefore, in the most sensitive part of the growth response curve. Additionally reductions in biomass in close proximity to the source is much less dramatic. At these locations, internal phosphorus levels are high (approx. 40 percent). Reductions in phosphorus loading will alter the internal phosphorus levels of the algae, but in most cases, pool levels will remain on the saturated (insensitive) part of the growth response curve (see Figure 3).

We have learned from examining this relatively small environmental perturbation at Harbor Beach that such events may drastically affect the capacity of the organism to respond to modest incremental improvements in water quality. From model projections it can be established that significant reductions in biomass may require almost complete removal of phosphorus, particularly near the source. Loading reductions of the magnitude necessary for a return to unperturbed conditions may approach the limit of cost/benefit analysis feasibility.

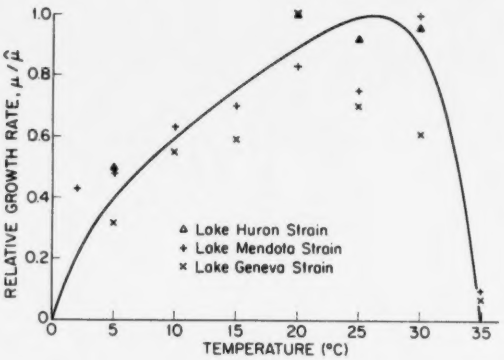

Figure 9. — Model projections of *Cladophora* biomass at various locations at the study site related to several levels of phosphorus removal.

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ROLES OF MATERIALS EXPORTED BY RIVERS INTO RESERVOIRS IN THE NUTRITION OF CLADOCERAN ZOOPLANKTON

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ABSTRACT

The differences between natural lakes and reservoirs are variously significant to the regulation of limnetic processes. Large suspended sediment loads 0.1 to 0.5 gms per liter are common in rivers of agricultural landscapes of the Great Plains of the United States. This material often reduces light penetration to the extent that photosynthetic production is inhibited, despite high nutrient loads that are also common in the Central Plains States. Such bodies of water do not fit most phosphorous-chlorophyll regression models; they are "low and to the right." Nevertheless, zooplankton populations do not seem to be reduced under such conditions. They are able to filter small particles and clays and use the bacteria and/or the dissolved organic matter associated with them. This implies that the river and its drainage basin continue to be the driving variable for secondary production in a reservoir in a fashion distinctly separate from natural lakes where zooplankton phenomena have been investigated more fully.

The differences between natural lakes and reservoirs are variously significant to the regulation of limnetic processes. (Neel, 1962; Baxter, 1977; Marzolf, 1980). Large loads of suspended sediments (0.1 to 0.5 gm per liter) are common in rivers of agricultural landscapes of the Great Plains of the United States. This material reduces light penetration, often to the extent that photosynthetic production is inhibited despite high nutrient loads also common in the Central Plains States, (Marzolf and Osborne, 1971). Such bodies of water do not fit most phosphorus-chlorophyll regression models; they are "low to the right" (Jones and Bachman, 1978). Nevertheless, zooplankton populations do not seem to be reduced under such conditions. This implies that the river and its drainage basin continue to be the driving variable for secondary production even under the lake-like conditions of the reservoir. The distribution pattern is distinct from natural lakes where zooplankton have been investigated more fully.

In this presentation we address the question: In the face of reduced photosynthetic production by algae because of silt and clay turbidity, what is the food resource of the filter-feeding limnetic zooplankton?

The alternative candidates for zooplankton foods are organic matter produced upstream in the watershed or in the river itself, and/or the bacteria that are decomposing this allochthonous material. Organic detritus enters the reservoir from the river as both dissolved and particulate fractions; the dissolved fraction is usually of greater mass, often by as much as 20 times. This has been reported from natural lakes and streams (Wetzel and Rich, 1973), from the oceans (Durrsma, 1960) and from a few rivers (Weber and Moore, 1967). The particulate fraction is directly available to filter-feeders; the dissolved fraction is not. Marzolf (1980) demonstrated in a preliminary way that dissolved organic matter can be rendered available to filter-feeders through adsorption onto clays, i.e., a dissolved amino acid adsorbed on clay was desorbed and retained by *Daphnia pulex* upon being allowed to filter such a suspension. It is not clear that dissolved organic matter generally can provide the nutrition to maintain zooplankton metabolism, growth, and reproduction by this mechanism since the nutritional qualities are likely to be variable, and in some cases, inadequate. Bacterial use of dissolved organic substrates, on the other hand, can render dissolved fractions particulate; with the incorporation of inorganic nutrients from ambient water the quality of the particulate organic matter as zooplankton food will increase. The details of that process remain to be demonstrated, but it is clear that the presence of clay particles enhances the activity of bacteria (Jannasch and Pritchard, 1972). The association of silt and clay particles, dissolved organic matter, and microorganisms offers a usable food resource.

We are not prepared to discount the continued use of algal cells as cladoceran food in turbid reservoirs but we consider their importance to be reduced for two reasons: (1) The largest concentrations of chlorophyll-bearing cells are found in the inflowing river water along with the highest concentrations of silt and clay particles. We show here that high clay concentration inhibits the feeding rate on algal cells. (2) Algal density and the rate of photosynthetic production are reduced with increasing distance from the inflow (Marzolf and Osborne, 1971); thus, just as the inhibitory effect of clay particles on filter-feeding is removed the availability of algal cells in the resource is diminished.

As part of an investigation into the roles of suspended silts and clays in zooplankton nutrition, we have made several measurements in the laboratory to document the ingestion of inorganic silt and clay particles by *Daphnia* sp. and the inhibition of algal ingestion in the presence of inert particles. The following describes this evidence.

METHODS

The first experiment reported here estimates the clearance and ingestion rate of clay particles by the cladoceran *Daphnia pulex* cultured from Tuttle Creek Reservoir. The experimental methods have been reviewed by Rigler (1971). The animals grazed, for at least 1 hour, in a suspension of non-radioactive particulate food to acclimate to the experimental conditions. The animals then were transferred to an identical, but radioactively labeled feeding suspension for 5 minutes, during which radioactive particles were ingested. This length of time was short enough that radioactive feces are not likely to be produced and long enough to minimize any effect of transferring the animals. After measuring the radioactivity of feeding suspensions and animals, clearance and ingestion rates were calculated (Rigler, 1971).

The coarse clay mineral particles (mean diameter is 4.65 micrometers) used in the clay ingestion experiment were processed from natural lake sediments. The dominant mineral was montmorillonite; illite was also present. After air-drying the sediments, they were roller-milled then pin milled into a dry, textured ground mineral composed of variously sized particles. Preliminary size fractionation of the milled sediments in a Bahco Micro-Particle Classifier (Harry Dietert Co.) was followed by wet fractionation with centrifugation. The suspended clay particles were labeled with the radionuclide Zn-65. This divalent cation adsorbs to the surface of the clay particles (Bachman, 1961), thus making possible the estimates of clearance and ingestion. In this experiment, clearance and ingestion rates were measured over a range of clay particle concentrations from 10^3 to 10^6 particles/ml. Four adult and four juvenile *Daphnia pulex* were used in each treatment: 25 ml of the desired concentrations of clay particles in DM2 medium (D'Agostino and Provasoli, 1972). The adult Daphnia were individually counted, while the juveniles were paired. Animals and filtered aliquots of suspensions were directly counted (Beckman 4000 Gamma Counter).

The second experiment examined the interference of algal ingestion by *Daphnia pulex* when suspended sediments were present. The alga, *Ankistrodesmus falcatus* var. *acicularis* was labeled by incubation with C-14 sodium bicarbonate. Feeding suspensions of labeled and unlabeled algae were prepared by adding the cells and the appropriate amount of washed lake sediments to the DM2 medium to produce the desired final concentration of both algae and sediments. These data are part of a larger experiment in which the concentrations of algae and sediments were varied. Four levels of algae (1.65×10^3 to 4.46×10^4 cells/ml) and six levels of sediments (0.0 to 160.0 mg/l) were used. Four adult *Daphnia pulex* were used in each treatment combination in 25 ml of feeding suspension. Prior to liquid scintillation counting, the animals were allowed to dry in open scintillation vials before adding a tissue-dissolving agent.

RESULT

Over the range of clay concentrations used in the clay feeding experiment, ingestion rates (Fig. 1) of both adult and juvenile *Daphnia pulex* increased with particle concentration (treatment effect of particle concentration on log ingestion rates: $P > F$ 0.0001 for both sizes). Clearance rates also declined as particle concentration increased ($P > F$ 0.0001 for adults, $P > F$ 0.0012 for juveniles). The slopes of the adult and juvenile lines within each parameter differed (log clearance rate: $P > F$ 0.0001; log ingestion rate: $P > F$ 0.0001) This means that the adult and juvenile animals used in this experiment differed in their response to increasing particle concentration. The adults ingested more particles as particle concentration increased and their clearance rates decreased less rapidly than did the juveniles.

The linearity of the ingestion rate function demonstrates that up to 1.0×10^6 coarse clay particles/ml, the capacity of these *Daphnia* to ingest these clay particles is not limited. Clearance rates are generally thought to be constant and maximal when particle concentration is below some saturating level (Hall, et al. 1976). This experiment suggests that clearance rates decline as ingestion rates increase in response to increasing particle concentration.

Ingestion of *Ankistrodesmus* cells by *Daphnia pulex* (Figure 2) is decreased by the presence of suspended sediments (treatment effect of sediment concentration on log ingestion rate: $P > F$ 0.0001) At a sediment concentration of 160 mg/l ingestion rates are reduced to about 6 percent of the rates in the treatments lacking sediments. Clearance rates also decline ($P > F$ 0.0001) as sediment concentration increases. Clearance rates represent the minimum volume of water that must be filtered to produce the observed radioactive disintegrations in each animal. The measure probably underestimates true filtering rates and it says little about the efficiency of filtration of algal cells or sediment particles as they are affected by sediment concentrations.

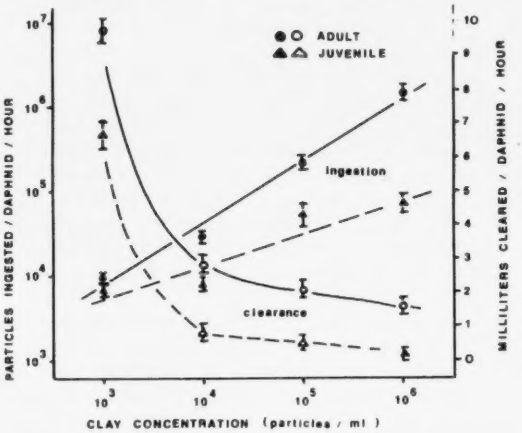

Figure 1. — Clearance and ingestion rates of *Daphnia pulex* in suspensions of coarse clay particles. Each point is the mean of 4 observations (adults) or 2 observations (juveniles), with standard error bars.

DISCUSSION

In this conference session devoted to the factors influencing the dynamics of eutrophication where most attention is given to nutrient responses of limnetic flora we are hesitant to divert too much attention toward processes centering on secondary production in reservoirs or suspended sediments. These subjects may be trivial in comparison to nutrient regulation of primary productivity in lakes and we are not inclined to say much to the contrary. Lake restoration projects, however, are often developed for bodies of water that are: (1) Artificial impoundments, (2) identified as turbid (often wrongly or simplistically associated with the eutrophic condition), and (3) not essentially regulated by the loading of the plant nutrients. It seems appropriate to discuss other biological phenomena that occur in lakes that need restoration whether they are eutrophic in the classic limnological sense or not. Our point of view is that the water quality in lakes, reservoirs, and streams is controlled to a large degree by biological processes. Further, that the organisms whose metabolism and activities are central to the control adapt to the physical and chemical environment influenced by that control.

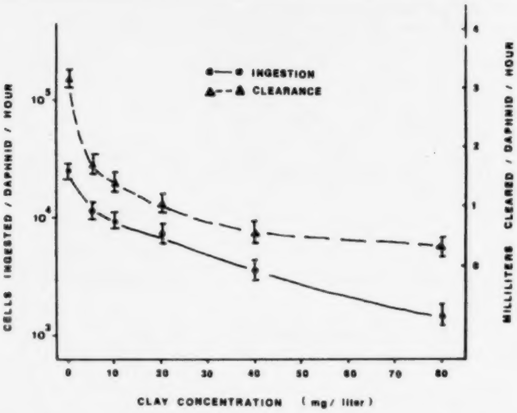

Figure 2. — Clearance and ingestion rates of *Daphnia pulex* in suspensions of *Ankistrodesmus* and clay sediments. Each point is the mean of 16 animals, 4 from each of 4 treatment levels of algal cell concentration (from 1.65 X 10^3 to 4.46 x 10^4 cells/ml), with standard error bars.

We have considered the concentration and quality of zooplankton resources to be of prime importance in explaining zooplankton activity. This is parallel to the perspective of our colleagues on this panel as they have considered nutrients and phytoplankton activity. The only difference is that we are constrained to focus on particulate materials because the resources of these filter-feeders do not include dissolved materials. In fact, a great unknown in evaluating the filter-feeding process is the lower size limit of usable particles. Is there a sharp threshold identified by the geometry of zooplanktonic filtering appendages? Do species of zooplankton differ in their capacity to use particulate resources at the small end of the size spectrum? Do zooplankters adapt to changing size frequency distributions of their resources by altering their filtering behavior?

It is our thesis that different species of cladoceran filter feeders respond differently to such changes in resource size frequency.

If this is true then we should expect to find some species at a competitive advantage in an environment where the dominant size frequency categories are small, say, less than 2 micrometers. We further suggest that the mechanism for resource availability that is related to the available surface area for the adsorbtion of dissolved organic matter will reward filter-feeders for their capacity to filter smaller particles. That is, the surface area per "gut full" of particles increases geometrically with decreasing particle size (Arruda, 1980).

SUMMARY

1. The impoundment of rivers to form reservoirs provides habitat for zooplankton in regions where natural lakes are rare.
2. The river and its drainage export silts and clays into the reservoir that regulate the trophic patterns in reservoirs by establishing gradients in turbidity and thus photosynthetic production.
3. Clay particles interfere with the filtration of algal cells by *Daphnia* and decrease ingestion of them as clay concentration increases.
4. Clay particles are ingested by filter-feeding zooplankton and may be usable as a food resource because of adsorbed organics and bacteria associated with them.

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METHODS OF ASSESSING NUTRIENT LOADING

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ABSTRACT

The practical application of measuring techniques published by various authors which quantify the nutrient load of a lake via influx and settlements have not been given sufficient attention. The investigation of the influx, although time-consuming and costly, is essential for lake budgeting and lake restoration studies. Experience acquired in connection with the OECD Eutrophication Program is passed on for future practical use. Along with a short literature review and an introduction to the theoretical correlation between waterflux and substance-concentration, various possibilities of measuring the probable load are stated. In addition to continuous samples proportional to the water flow in the nutrient-rich influxes (more than 15 percent of total annual load), random sampling of other influxes (more than 5 percent of total annual load) is recommended. The waterflux of the most important in- and outflow must be registered automatically. High water demands special attention. Those areas not covered by direct measurements can be assessed by statistical methods. The problem of nutrient export coefficients is briefly handled. At the end, recommendations for practical application are given.

INTRODUCTION

Traditionally, limnological research is based on in-lake investigation, and loading measurements have often been neglected. However, in the past 5 to 10 years, increasing attention has been given to this problem. Vollenweider (1968) was the first to establish a quantitative correlation between nutrient loading and the trophic state of a lake, thereby enabling us to quantify the biological response of lakes. Application of such models requires, however, a sound knowledge of the component balance, which can only be achieved by an admittedly more complex inlet and outlet investigation.

The fundamental objective of the recently completed OECD Eutrophication Program (Fricker, 1980), was to study the correlation between nutrient loading of lakes and their biological-chemical (trophic) response. Another aim was to determine a critical loading value, i.e. a level which can be tolerated by the lake without changing its trophic character.

A preliminary evaluation showed that in many cases, the influxes (e.g. measurement of water flow, flood surveillance) had not been sufficiently investigated as initially planned.

The sum of all nutrient inputs is the decisive factor controlling the trophic state of a lake. Quantitative evaluation of the nutrient sources is of prime importance for planning purposes, also for assessing the effectiveness of water pollution control measures in the catchment area and in the lake itself.

The following are the most important sources of pollution:

- Point sources:
  input via influents
  input via domestic wastewater from sewer systems
  wastewater treatment plant effluents
- Diffuse sources:
  input via precipitation
  urban and agricultural runoff
  groundwater
  lake sediments

When planning an influx investigation, it is essential to assess in a preliminary study, the loading fraction of each influx while always taking into account the water flow; furthermore, the measurement must be limited to the most important inflows. These should constitute at least 80 to 90 percent of the total load. Here, it must be taken into consideration that an investigation of a minor influx requires the same technical and analytical sampling procedure as a major influx.

There have been only a few examples of an analytical classification of the pollutant loads in lakes: Ambühl (1979), Lohri (1977), Treunert, et al. (1974), Unger (1970), Wagner (1969), Wagner, et al. (1976). On the other hand, several studies have been carried out on the monitoring of river quality (Dandy and Moore, 1979; Davis and Zobrist, 1978; Manczak, 1968; Manczak and Florczyk, 1971; Sanders and Adrian, 1978; Stevens and Smith, 1978) which deal with the problems of outlet and nutrient concentration measurements, as well as with the assessment of nutrient loads.

The aim of this paper is to give a brief review of the theoretical background between water flow and nutrient concentration, and to discuss the different ways to measure nutrient load, including the most important factors involved.

RELATIONSHIPS BETWEEN WATER FLOW AND NUTRIENT CONCENTRATION

Because of the special conditions prevailing in the catchment area, a fairly close correlation exists in every water body between water flow and the simultaneously measured concentration of a pollutant. This correlation can also be used in determining pollutant loads (Manczak, 1968; Manczak and Florczyk, 1971). According to Bernhardt, et al. (1974), such a correlation does not seem to exist in small to middle-sized rivers (e.g. Wahnbach approx. 1 m^3/s) with low water flows. However, they obtained similar results as Manczak and Florczyk (1971) with higher water flows. These authors proposed three types of correlation (Figure 1) based on two overlapping processes: (a) dilution, a low concentration with an increasing water quantity, and (b) erosion, an increased concentration with a higher water flow. The shapes of the curves in Figure 1 depend on many factors. The most important are:

1. Chemical components.
2. Degree of pollution in the river.
3. Its hydrotechnical and hydrological characteristics.
4. Its self-purification capacity (temperature dependent).
5. Distance of the monitoring station from any source of pollution.
6. Quality and quantity variation of the discharged loads.

In heavily polluted rivers (Type I), the main factor influencing the shape of the curve is the dilution of wastewater with increasing water runoff. In clean rivers (Type II), the concentration increases with increasing water runoff as a result of greater resuspension and transport of river sediments. In intermediately polluted rivers with a low flow, the dilution of wastes is the predominant factor, so that the curve is similar to Type I. At higher flows, the dilution effect disappears and the influence of resuspension of bottom deposits and washout of the drainage area dominates. Thus, with flows which are higher than the annual average, pollution increases with a higher flow. The concentration can decrease again when diluted with even higher water quantities. Based on this model, analytical classification concepts of the total load into different sources (loading from wastewater, runoff, ...) are presented by various authors (Dandy and Moore, 1979; Davis and Zobrist, 1978; Liebetrau, 1979; Manczak and Florczyk, 1971; Smith, 1977; Zobrist, et al. 1977).

To start a lake balance investigation and loading model, the annual load should first be determined. However, for further evaluation of water protection measures in a lake and catchment area or for developing specific lake restoration strategies, it is essential to quantify the different sources of pollution. In practice, variation of the analytical values of the correlation between water flow and concentration is best compensated by the function of a higher degree. According to Wagner (1969), polynomials proved to be suitable, in particular the equation $Y = a/x + b + cx + dx^2 + ex^3$, as they often supply the least squares sum, and are also comparatively easy to calculate. Polynomials are often derivatives of the common type:

$$Y = \sum_{i=-1}^{3}(B_i \cdot X^i)$$

Y = material conc.; x = amount of flow
B_i = regression coefficient
i = exponents and indices of the regression coefficents
(according to Wagner, et al. 1976)

Depending on the sample taking technique (random, collective samples), an adapted polynomial function of concentration and water flow neglects the seasonal and/or daily concentration variations. According to Davis (1980), the often observed large differences in concentrations at the same water flow are not necessarily caused by deviation, but rather by seasonal fluctuation. For example, in the case of nitrate (or phosphate) it is mainly the water temperature and resulting biological activity.

McMichael and Hunter (1972) and Thomann (1967) have introduced a cosine function to account for both annual and weekly cycles in the flow model. Bührer (in preparation) combines the polynomial of the regression computation with a Fourier series over time. Schweingruber (1980) has tried to establish a linear combination between the annual cycles (sine function) and the water flow dependent (polynomial) terms, but optimization of all coefficients proved to be difficult. Basically, the calculation procedure for assessing loads must be improved. Until new models are tested on different lakes, the polynomial remains the most suitable function for practical application. Above all, no correlation has yet been established between the computed results from integrated collective samples and the ones obtained from random samples of the same river. Besides, the nutrient loads registered during 1 year may not be transferred so readily from 1 year to another. Particularly the varying climatic data should be taken into consideration.

PRACTICAL CONSIDERATIONS

An influx investigation program should cover the concentration of water substances over the entire range of the water flow in order to statistically guarantee the load calculation.

Numerous individual analyses are technically always possible in low and normal water levels. In high water, however, where individual results can fluctuate considerably, special measures should be undertaken. According to Keller (1970), the total phosphorus load remains small up to approximately five times the mean annual flow. It increases only with high waters which contain at least 5 to 10 times the mean annual flow, thereby increasing particulate organic phosphorus, while dissolved phosphorus remains small. Unger's investigation (1970) of the Argen (mean water flow 18.6 m^3/s) shows that, in only 9 days, 10 percent of the annual water volume, 25 percent of the total phosphorus, and 15 percent of the phosphate load flowed off. In general, not only do high waters significantly increase substance flow off, but also momentary water level increases after dry periods. Therefore, as expected, greater component volumes are discharged with increasing water levels than with decreasing water levels directly after peak water levels.

How often do high waters occur? Should data of the mean daily flow values exist, the occurrence probability may then be assessed by a frequency distribution. Depending on the sampling method used, additional samples must be taken after and during abundant rainfalls. A critical value of 15 mm rainfall/day (Wagner, et el. 1976) was registered for Lake Constance. Nevertheless, it will never be possible to determine the nutrient load of the high water maximum for each river. Already the requirement, to determine as often as possible, unforseeable high water flux, demands numerous personnel. Generally, the extrapolation range between the extreme high-water situation, where no concentration measurements exist, and the range in which concentration measurements exist must be minimized. Polynomes used as fitting-curves tend to deviate too strongly beyond the plotted values. This can cause large errors in the load calculations.

MEASUREMENT OF WATER FLOW

The accuracy of nutrient load budgeting depends upon the accuracy with which the water flow of the influx is measured. According to Bernhardt, et al. (1974), continuous water flow measurement is a factor two to four times more accurate than intermittent measurement. Consequently, it is somewhat absurd to determine a substance concentration with great precision and only estimate the water flow roughly. The exact determination of the water volume requires a calibrated sampling spot in the flowing water, with a solid installation safe from high waters, and equipped with a water-level-registering instrument (limnigraph, water gauging station). The water volume can then be calculated from the water level by two methods:

A. A water-velocity-profile is taken of the stream. The velocity, multiplied by the cross-section area, gives the water volume per unit time. When this is done for various water levels, probable relations between water volume and water level can be calculated.

B. In the literature (Weyrauch, 1915), empirical relations are described for measuring weirs with known profiles but apply only for slow-moving water currents.

MEASUREMENT OF CONCENTRATION

Different methods of sampling running waters exist (Wagner, 1980; Ambuhl, 1973): Continuous sampling (automated) over several days, 24-hour mixed samples, a combination of several random samples to 24-hour mixed samples and single samples. These methods must be applied according to the loading importance of the different flowing waters, which can be estimated by the mentioned pre-study. An example here is the Lake of Sempach study (EAWAG, 1979) in which all influxes were walked off, once in wet weather and once in dry weather. The annual load was approximated on the basis of the relative frequency of the weather condition in that the dry weather study was weighted with a factor 5 and the wet weather study with the factor 1. All influxes whose yearly load supply more than 5 percent of the calculated total load are considered important.

SINGLE SAMPLES

Waters whose load constitute 5 to 15 percent of the total load of a substance can be accurately measured by random samples, providing they are not subjected to systematic fluctuations and have a relatively constant water flow, even at times of precipitation. To minimize the high costs of river quality monitoring, Sanders and Adrian (1978) have developed special statistical criteria based on the standard deviation formula. According to Bernhardt, et al. (1974) the standard deviation of the load, determined from a theoretical reference value, is 20 percent for a sampling frequency of 28 days, and 5 to 10 percent for 14 days. The sampling days must be distributed over the investigation period according to a specific time plan. For example, the influxes of Lake Constance (Bodensee) are investigated regularly all 18 days (not a factor of 7); therefore on each weekday the influxes are investigated three times during the year (Wagner, et al. 1976). The sampling must also include all hydrological conditions and be supplemented with sampling during high water. It is best when the water volume is registered continuously or at least daily. This way the random sample method requires the least effort, and the load error can be determined statistically. Diurnal fluctuations, depending upon the model function, can be taken into account. In sewage-loaded streams, the day-night rhythm of sewage-originated substance concentration must be included in the calculation. This can be taken into account by using a complementary investigation program (EAWAG,1979): First, the sampling times of all random samples must be known. Secondly, during a period of normal water flow, a 24-hour continuous sample is taken, and during the same time, many single probes are taken in short intervals. From the ratio of the single probe concentration and the 24-hour mean concentration, factors are calculated to correct the measured random sample concentrations to a mean daily value.

24-HOUR CONTINUOUS SAMPLES

In addition to the regular random samples, 24-hour continuous samples are recommended. Instruments for this use are already on the market. They take water samples proportional to the water level over a longer period of time (hours to days). The sampler of Quantum Science Ltd., England is basically a plastic cylinder which is anchored directly in the flowing water. The amount of water entering the cylinder is regulated by an adjustable valve which controls the amount of air escaping from the cylinder. The analytical data obtained in this manner can be treated statistically the same as the random samples. The measured concentration is set in relation to the mean water flow over the same time period (limnigraph). In this manner, the problem of diurnal fluctuations can easily be avoided.

WATERFLOW-PROPORTIONAL PERMANENT SAMPLING

An influx whose nutrient load constitutes an important part (more than 15 percent) of the total load, must be taken with permanent sampling devices. Mountain streams, which can overflow rapidly during storms and thereby transport a significant portion of the total load into the lake within a short time also belong to this category of influx. Fraction samples are taken which are proportional to a previously determined water volume, i.e., the greater the water flow, the greater the frequency with which fractional samples are taken. Optimal sampling results when the sampling is regulated by a water gauge (limnigraph). This implies that the relationship between water level and water flow must be accurately known in advance. Correct operation of the installation depends on the exactness of this relationship. Depending on the amount of samples taken, problems can arise in high water situations, so that, in spite of increasing water flow, the sampler cannot operate quicker. For this reason, the automat must provide for at least two ranges of water flow, each of which applies for a certain ratio between fraction sample and total water volume. For each range a separate sampling container is necessary. The easiest method of conserving the probes is in a refrigerator. The accuracy of determining the nitrite and ammonium concentration is lessened with the duration of stay. The weekly analyzed sample concentrations are multiplied by the corresponding water flow and added up week by week for the total load.

This method is surely the most reliable for determining the total load and compared with the random sampling method, mathematically much less complicated to handle. The division of the total load into its origin is not possible by mathematical techniques; here additional random samples are necessary.

OTHER SOURCES

Sewage treatment plants: The concentration and load of nutrients at the outlet of a sewage treatment plant can be described only poorly, even with the help of complicated functions. The total daily load can be measured accurately by using a suitable collector device; several 24-hour integrated samples would be sufficient. Weekly and seasonal variation should be taken into account. Synchronous measurement of the flow is essential. In the evaluation, differences caused by rain water overflow are to be taken into account.

Diffuse sources: Most authors (e.g. Duncan and Rzoska, 1979; Uttomark, et al. 1974) agree that increasing land use and the substantial outflow of particulate material from the drainage area results in serious deterioration of many water bodies. The kinetic energy of flowing water is the primary transport mechanism of these materials. Other influencing factors are: General topography, contour, soil properties, vegetative cover, agricultural practices and livestock, and precipitation. While further research on the cycle of nutrients is certainly necessary (c.f. MAB project No. 5: Mechanisms of land use impacts on inland waters), effective means of reducing agricultural input into receiving waters are already known but often, at least from an economical point of view, difficult to realize. Origin of nutrient input which cannot be analytically determined (e.g. single housing on the lakeshore, ground water from slopes, areas outside of the drainage area of investigated flowing waters (statistical area) must be estimated. This can be done with the help of so-called nutrient export coefficients found in the literature. Another possibility is to set the unknown nutrient export of an area proportional to that of a neighboring area whose export has been analytically determined.

NUTRIENT EXPORT COEFFICIENTS

It is quite probable that enough basic data are available for some catchment areas to calculate the nutrient load. In general, applying nutrient export coefficients from literature may be sufficient to plan an investigation program, even though, with regard to accuracy, their values are not exact. There is a substantial quantity of literature (see review in Uttormark, 1974) for estimating the nutrient input into lakes. It is apparent from the data that considerable variation exists in the quantity of nutrients that are exported from similar areas devoted to the same use. Latest research from Greifensee studies has shown that compared to the values found in the literature (calculations on the basis of seven test areas; Gächter and Furrer, 1972) the phosphorus export coefficients can be up to twice that amount. In practice, we recommend (in accordance with MAB 5) that in every large influx study, a small test area be investigated. Here, land use should be defined in some detail as to type of agriculture, forestry, recreation, and tourism, and the intensity of usage should be quantified (e.g., fertilization rates). The nutrient export from this area must be intensively studied. This will produce representative data so indispensable to the calculation of load in the so-called statistical drainage area (area with no loading measurements).

CONCLUSIONS AND RECOMMENDATIONS FOR PRACTICAL APPLICATION

Studies of the influx are vital for lake restoration programs. A pre-study can reduce the investigation program to the most important (nutrient rich) influxes. The waterflux must be measured at least once per day, or better even, continuously registered with a limnigraph. The influx with a nutrient load of more than 15 percent of the total load should be investigated with a waterflux proportional sampling automat (7 days a week). To be accurate, the relationship betwen waterflux and water level must be given special attention by repeated measurings. For influx with loads up to 15 percent, a 1-day continuous sample or a random sample (e.g., once every 18 days) is sufficient. To obtain a statistically sufficient distribution of samples over the entire range of water levels, including high waters on call, special planning is required. To evaluate random samples, a polynominal regression is recommended, even though the systematical concentration fluctuations are not taken into account by this method. Improvements in this direction must be attentively followed! Normally, for sewage treatment plants, several 24-hour continuous samples are sufficient. Depending on the conditions, precipitation analysis may or may not be disregarded. The nutrient export for parts of the drainage areas which have not been measured, can be calculated by export coefficient values found in literature. Better yet is the application of export coefficients acquired from special test areas.

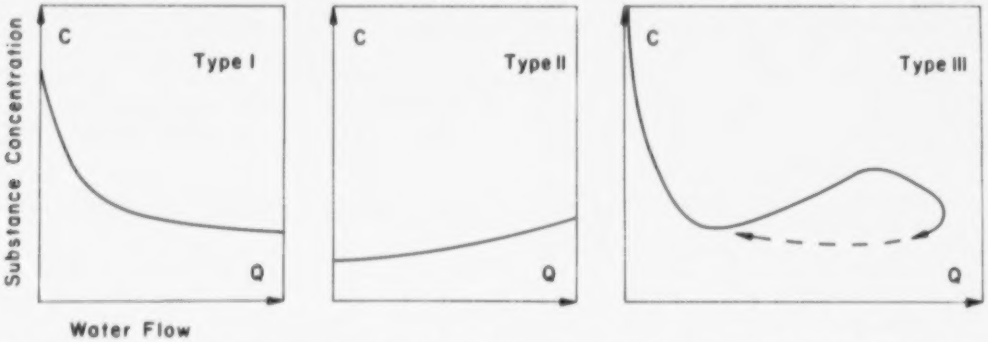

Figure 1. — Basic types of curves representing the correlation between concentration of pollutants and rate of flow (Manczak and Florczyk, 1971; Wagner, et al. 1976).

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QUANTIFICATION OF PHOSPHORUS INPUT TO LAKES AND ITS IMPACT ON TROPHIC CONDITIONS

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ABSTRACT

A simple model for quantification of nonpoint sources of pollution for a watershed is presented. Analyses of sampled data from the Lake Waramaug and Still River watersheds are used to validate the accuracy of the model. It can also be applied to develop a cost-effective watershed control management plan. A discussion of the role of decision analysis techniques in lake management is presented in a conceptual form.

Under the Clean Water Act, Congress set an interim (July 1983) national goal of achieving, wherever attainable, water quality which provides for the protection and propagation of fish, shellfish, and wildlife, and provides for recreation in and on the water. The 1977 Amendments of the Act emphasized restoring lakes with potentials for significant recreational usage. A recent report by the Council on Environmental Quality (1979) indicates that 67 percent of the lakes in the Nation may have serious water quality problems. Under the Clean Lakes Program of the Act, Federal assistance is available to develop and implement lake management plans.

To provide an effective lake management plan, a nutrient budget must be produced. Phosphorus, being the most manageable, is usually the nutrient considered in lake management planning studies. To be useful for planning purposes, the phosphorus budget needs not only to detail the net inflows and outflows, but also to identify sources, their location and magnitude. These details are needed to develop a cost-effective lake watershed management plan. The Clean Lakes Program fully recognizes this need. The diagnostic/feasibility studies that must be conducted prior to the implementation of lake restoration measures, require, in addition to other information, a description of land use and an assessment of the role of point and nonpoint sources of water pollution within the watershed.

While quantification of point sources is a routine procedure, the inherent temporal and spatial variability in nonpoint sources presents serious obstacles in their quantification. Difficulty in quantifying nonpoint sources is probably one of the primary reasons for failure of all but a few of the 208 studies to effectively deal with the problem of nonpoint sources.

Sampling programs aimed at obtaining reliable estimates of nonpoint sources could be very expensive, especially if one is interested in identifying the location and quantity of major sources within the watershed. More sophisticated computer based models often require significant quantities of data for calibration and verification purposes. Using them on a routine basis is often beyond the means and resources of a planning agency. On the other extreme, simple areal loads, i.e., an average emission rate for each type of land use, do not allow for adjustments reflecting the variability in site specific values of causative factors in the watershed.

Over the last few years, The Center for the Environment and Man, Inc. has formulated a model for quantification of phosphorus and other pollutants from nonpoint sources to lakes and streams. To date, the model has been used, in one form or other, in developing lake management plans for 16 lakes in Massachusetts and Connecticut. In addition, the model is being used by the State of Connecticut to assess the impact of nonpoint sources in 94 watersheds.

MODEL DESCRIPTION

The model developed by CEM for Computing Loading Estimates from Nonpoint Sources in a watershed (CLENS) is a simple model with data requirements limited to those which are readily available, at least for most of the Eastern States. The model differs from areal load models as it allows for including site specific information. In contrast to more sophisticated nonpoint source models, its resource requirement is modest. The model can be considered more comprehensive as it includes computation of nonpoint pollution from sources other than erosion. In all, seven specific sources of nonpoint pollution are considered in CLENS. These are:

1. Washoff from urban areas.
2. Erosion from other areas.
3. Washoff from barnyards and feedlots.
4. Leachate from landfills.
5. Washoff from roadways.
6. Leachate from septic systems.
7. Wet and dry fallout.

The basic formulations for each of these sources have been derived from several EPA sponsored studies (Midwest Res. Inst. 1975; Heaney, et al. 1977; Shaheen, 1975; U.S. EPA, 1976, 1977) and studies by the Agricultural Research Service (Wischmeier, 1960, 1976; and Soil Conserv. Serv. 1976). Specific formulations and equations for each of the seven sources have been described in detail elsewhere (Ahmed, 1979; Ahmed and Schiller, 1980). CLENS is a management type model; in that respect it does not follow the standard protocol of modeling, i.e., calibration and verification prior to simulation. Instead, the values of the parameters used in the model are obtained from available data, reports, and studies.

The parameters utilized in the model can be placed into two categories — parameters whose values are valid for a region (one or more watersheds) and parameters whose values are chosen for a specific location in the watershed. Examples of regional parameters are crop management factors, leachate characteristics, and pollutant loadings at roadways. Examples of parameters with site specific values are population density, soil type, and traffic density. A favorable comparison of nonpoint source loads computed by CLENS and those estimated from field programs at specific locations (as described later) in Connecticut indicate that such an approach is feasible.

The overall purpose of the CLENS model is to provide water quality and land use planners with a tool to construct preliminary nonpoint source pollutant budgets. Because the nonpoint source loads computed by CLENS can be associated with specific areas of the watershed, they provide the needed versatility to allow planners to create cost-effective management plans.

In addition to quantification of nutrients, organics, and sediments, CLENS can also be used for:

- Analysis of tradeoff between advanced wastewater treatment and nonpoint source control.
- Evaluating the effectiveness of best management practices.
- Locating and quantifying nonpoint sources from "hot spots."
- Designing an efficient sampling program.

ASSESSMENT OF TROPHIC CONDITIONS

An estimate of the expected quantities of phosphorus loads alone does not indicate the severity of the water quality problems in a lake. Development of lake management plans must be based upon consideration of the impact of phosphorus loads into a lake. Hence, the estimate of annual phosphorus loads is translated into expected average in-lake phosphorus concentrations through use of lake models such as the one proposed by Dillon and Rigler (1974). The Dillon-Rigler model can be expressed as:

$$P = \frac{(P) \cdot Q}{1 - R_p}$$

in which:

P = total annual phosphorus load to lake (g/yr).
Q = total annual outflow from the lake (m/yr).
$[P]$ = mean annual outflow phosphorus concentration (g/m).
R_p = phosphorus retention coefficient.

The phosphorus retention coefficient can be computed using the relationship developed by Dillon and Kirchner (1975) which is:

$$R_p = \frac{V}{(V + q_s)}$$

in which:

V = net settling velocity (m/yr).
q_s = area water load (m/yr).

Based upon computed phosphorus concentration, lakes may be categorized by anticipated trophic status.

APPLICATION OF CLENS TO LAKE WARAMAUG AND THE STILL RIVER BASIN

So far, CLENS has been used to study 16 lakes and over 90 river watersheds. This paper presents the results using it on Lake Waramaug and the Still River Basin, both in Connecticut. These two examples have been selected for presentation because a modest amount of field data exists to assess the predictive accuracy of the model.

For application of the CLENS model to watersheds in Connecticut, the values of regional parameters were determined principally from a review of soil loss studies by the Soil Conservation Service and data available from the lake quality monitoring program of the Connecticut Department of Environmental Protection. The regional parameters once developed for Connecticut were applied to the Lake Waramaug watershed and the Still River Basin without any further modification. Watershed specific data such as population density land use, soil type, slope, traffic density, and other were obtained from available maps, reports and studies. It is noteworthy that the application of CLENS to large watersheds requires that watersheds be subdivided into subunits of homogenous land use and topographic conditions.

Lake Waramaug

Lake Waramaug is the second largest natural lake in Connecticut. The lake has a surface area of 2.7 square kilometers and a drainage area of 36.6 square kilometers. land usage within the basin is distributed as follows:

- Agriculture = 10 percent
- Forest = 66 percent
- Pasture = 4 percent
- Urban = 5 percent
- Wetlands/water bodies = 13 percent
- Recreation = 2 percent

Figure 1 shows Lake Waramaug and its watershed. CLENS was used to develop estimates of phosphorus loads from all sources within the watershed under pristine, current, and year 2000 conditions. The results are shown in Table 1.

From March 1977 through April 1978, the U.S. Geological Survey in cooperation with the Lake Waramaug Task Force and other local and State agencies conducted an extensive watershed and in-lake sampling program. The sampling data were analyzed to create flow-flux curves which in turn were used to develop estimates of net phosphorus and sediment export from the watershed. To truly compare the results obtained by the model and the sampling program, loads from other sources, such as septic tanks, were added to the loads estimated from the sampling program. Another estimate of the total

Table 1. — Estimated annual phosphorus loads — Lake Waramaug.

Time Frame and Source	Subbasin							Total	Percent
	A	B	C	D	E	F	Lake		
Pristine Conditions									
Erosion-Related	204	142	95	43	47	84		615	83
Atmosphere							130	130	17
Total	204	142	95	43	47	84	130	745	100
Consequent Lake Condition[2] = High Oligotrophic (Condition 1.8)									
Current Conditions									
Erosion-Related	445	234	260	128	60	260		1,387	73
Atmosphere							267	267	14
Septic Systems	5	25	68[3]	20		21		139	7
Landfills								0	0
Livestock	31					29	40[5]	100	5
Motor Vehicles	7	1	2			1		11	1
Point Sources								0	0
Total	488	260	330	148	60	311	307	1,904	100
Consequent Lake Condition = Mid-Eutrophic (Condition 3.1)									
Year 2000 Conditions									
Erosion-Related	511	326	416	143	71	382		1,849	73
Atmosphere							267	267	11
Septic Systems	92[4]	32	121[3]	30		45		320	12
Landfills								0	0
Livestock	31					29	40[5]	100	3
Motor Vehicles	7	1	2			1		11	1
Point Sources								0	0
Total	641	359	539	173	71	457	307	2,547	100
Consequent Lake Condition = Mid-Eutrophic (Condition 3.4)									

[1] As computed by CEM composite land use analysis.

[2] Lake Condition: 1.0 - Very oligotrophic
2.0 - Mesotrophic
3.0 - Eutrophic
4.0 - Hypereutrophic

[3] Includes 7 kg/yr from The Casino.

[4] Includes 87 kg/yr from Hopkins Inn and The Inn.

[5] Represents estimated annual P load from waterfowl.

Table 2. — Comparison of phosphorus loads — Lake Waramaug.

Computation Model	Computed Load (kg P/yr)
USGS Sampling Program	1,700
Dillon-Rigler Model	1,648
CLENS Model	1,904

Table 3. — Comparison of phosphorus loads — Still River.

Study	Computed Load (kg P/yr)
Frink's Study	61,820
NES Study	70,000
CLENS Model	66,770

phosphorus load was obtained from using the observed outflow phosphorus concentration data in the Dillon-Rigler model. Hydraulic characteristics of Lake Waramaug were used to determine the expected settling velocity in the lake. Table 2 presents the loading estimates for the three separate analyses.

The three loads are considered to compare favorably. It is noted that the CLENS model estimates the long-term average phosphorus loads while the other two estimates provide an estimate specific to the year of sampling.

Still River Basin

The Still River Basin is located in western Connecticut. The total area of the basin is approximately 184 square kilometers. The watershed (Figure 2) is rather steep and at the same time is heavily urbanized, with a population of 68,000 in the basin. The land uses within the basin are as follows:

- Residential = 32.5 percent
- Commercial = 3.1 percent
- Industrial = 4.5 percent
- Agricultural = 7.4 percent
- Forest = 36.6 percent
- Wetlands/water bodies = 15.0 percent
- Other = 0.9 percent

As part of the Connecticut 208 Program, CLENS was applied to the Still River Basin to compute annual nutrient and organic loadings.

The Still River is a tributary of the Housatonic River, entering it at Lake Lillinonah. A phosphorus budget for the Still River has been prepared by the U.S. EPA National Eutrophication Survey. In addition, Dr. Charles Frink of the Connecticut Agricultural Experiment Station developed a phosphorus budget for the Still River on the basis of data collected over a period of 1 year.

The total annual phosphorus loads for three separate analyses are presented in Table 3. The three values are comparable.

FORMULATION OF THE LAKE WATERSHED MANAGEMENT PLAN

CLENS can be used very successfully to analyze the cost-effectiveness of various phosphorus control measures in a lake watershed. The model has been applied in this mode to several lakes in Berkshire County, Mass. As an example, cost-effectiveness analyses of several phosphorus control alternatives for Lake Onota are shown in Table 4. Further details in cost-effectiveness analysis are available in the Upper Housatonic 208 Water Quality Management Plan (Berkshire County, 1977).

ROLE OF DECISION ANALYSIS IN LAKE MANAGEMENT PLANNING

To date, several lake models have been developed and several of them have been reviewed by Reckhow (1979). Most of the simple lake models share one basic characteristic — they are empirical models derived from statistical analyses of data for several lakes. While the models have a statistical basis, they are deterministic in the sense that they predict a fixed value of in-lake concentration.

Most recently, emphasis has been placed on the quantification of uncertainty associated with the predictions of in-lake phosphorus concentration (Reckhow, 1979; Reckhow and Chapra, 1979; Chapra and Reckhow, 1979). Reckhow (1979) considers the error term associated with phosphorus prediction to consist of model error, parameter error, and loading error. Reckhow and Chapra (1979) rightly point out that the non-negligible portion of the sum of the model error and parameter error may be due to phosphorus loading uncertainty in the model development data. Chapra and Reckhow (1979) analyzed the data from north temperate lakes to establish probability curves for a phosphorus prediction to fall within a particular trophic class.

From a planning point of view, the uncertainty in the loading estimates and the variation of loading from year to year are important. Review of sediment yield studies would show that the year-to-year variation could be significant. Hence, a decision about controls to improve the quality of the lake on the basis of a year's observation could be erroneous. Over the life of the project, the quality of lake water may substantially deviate from that expected in the planning analysis. This paper presents an approach for handling the variation loads over time.

For all practical purposes most of the lake models can be expressed as follows:

$$(P) = \frac{P\lambda}{V}\left(\frac{1}{1+\sqrt{\lambda}}\right)$$

in which:

λ = hydraulic detention time (yr).

V = lake volume (m^3).

For the sake of simplicity, if one assumes that the values of and V are constant over time, then a knowledge of the distribution of P could yield the distribution of [P]. However, in almost all cases, it would be difficult to identify the distribution of P, unless one entered into a multi-year sampling program. P, however, can also be expressed as follows: in which:

Q_i = annual inflow (m^3/yr)

P_i = influent concentration (g/m^3).

While not available directly, the distribution of P and Q can be identified based upon analyzing data from surrounding similar watersheds. It is noted that CLENS provides a long-term average value of P. If P and Q distributions are known, the distribution of P can be computed as follows:

$$F(P) = \int_{-\infty}^{P}\left[\int_{-\infty}^{\infty}\frac{1}{|P_i|}\cdot f(p_i \cdot \quad \frac{P}{P_i})dP_i\right] dP$$

Once the distribution of P is defined, the trophic status of the lake can also be defined in probabilistic terms.

If Q is a time dependent variable then the assumption of both and V as constant over time is contradictory. In reality, λ is a variable; however, consideration of as a variable poses difficulties in obtaining an analytical solution. Such a problem is, however, amenable to simulation. A simple simulation routine can be used to identify the distribution of the lake's trophic status.

APPLICATION TO LAKE MANAGEMENT

Improving lake quality enhances its use and benefits. It is reasonable to assume that the functional relationship between utility and the trophic status of a lake is not linear. In an actual planning setting, the concerned parties (lake associations or lake study committees) may produce this utility curve based on the resident's specific needs and desires. A hypothetical curve relating benefits and trophic index is shown in Figure 3. Having defined the distributions of the load and the trophic status, and the utility curve, decision-aid techniques, such as a payoff or decision tree, can be used to develop an optimum control plan. For illustrative purposes, a payoff table is shown in Table 5.

In Table 5, the value of benefits (b's) will be derived from a utility function. Based on the preceding analysis, a plan which maximizes the expected payoff should be expected.

SUMMARY AND CONCLUSIONS

The results of the detailed sampling program in the Lake Waramaug and Still River Basin watersheds validate the accuracy of the CLENS model. In addition to quantification of nonpoint sources, the model has been applied to develop cost-effective watershed management control alternatives. The model in its initial application can also be used to establish an effective field monitoring program.

Though the use of CLENS does not require a computer, the computer program developed at CEM substantially enhances the efficiency of its application.

While additional research is needed, it is apparent that the application of a statistical decision theory is quite appropriate to lake management planning and could provide support for the selection of plans providing maximum benefits over the long run.

Table 4. — Cost effectiveness analysis of lake watershed control measures.

Lake	Control Measure	Effectiveness (kg P/yr)	Cost ($1,000)	C/E ($/kg P/yr)	Remaining P (kg P/yr)	Lake Condition
Onota	1. Do nothing	--	--	--	1,269	2.7
	2. Manage manure	56	1.4	25	1,213	2.6
	3. Maintain catch basins	35	1.2	33	1,178	2.6
	4. Manage crops per SCS	17	1.5	88	1,161	2.5
	5. Sewer	236	62.0	260	925	2.3
	6. Control construction practices	30	18.9	630	895	2.3
	7. Build detention pond in Subbasin P	34	30.0	880	861	2.2
	Recommended Program	408	115.0	280	861	2.2
	Alternative Onota Program (without sewers):					
	1. Do nothing	--	--	--	1,269	2.7
	2. Use nonphosphorus detergents	84	0.8	10	1,185	2.6
	3. Manage manure	56	1.4	25	1,129	2.6
	4. Maintain catch basins	35	1.2	33	1,094	2.5
	5. Manage crops per SCS	17	1.5	88	1,077	2.5
	6. Manage septic systems	93	8.9	96	984	2.4
	7. Control construction practices	30	18.9	630	954	2.3
	8. Build detention pond in Subbasin P	34	30.0	880	920	2.3
	Alternative Program	349	62.7	180	920	2.3
	*9. Sewer	59	62.0	1,050		

* Not recomended because of poor cost effectiveness or insignificant change in lake condition.

Notes 1. Each control measure is evaluated as if the measures above it were operating. For example, the effectiveness of non-recommended sewering in this table is greatly reduced because much of the phosphorus will have been removed by the use of nonphosphorus detergents and the management of septic systems.

2. Lake Conditions: 1.0-1.9 = oligotrophic; 2.0-2.9 = mesotrophic; 3.0 and higher = eutrophic.

Table 5. — Payoff table for lake management plans.

Lake Status Phosphorus Contamination	Probability	Control Plan 1	Control Plan 2	Control Plan 3
$<(P_1)$	P_1	b_{11}	b_{12}	b_{13}
$(P_1)-(P_2)$	P_2	b_{21}	b_{22}	b_{23}
$(P_2)-(P_3)$	P_3	b_{31}	b_{32}	b_{33}
$>(P_3)$	P_4	b_{41}	b_{42}	b_{43}
Expected Payoff		$\Sigma P_i b_{i1}$	$\Sigma P_i b_{i2}$	$\Sigma P_i b_{i3}$

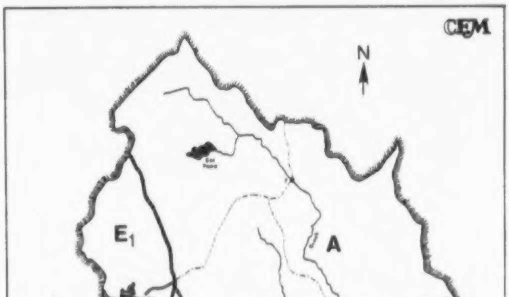
CEM
N
E1
E2
F
A
B
C
D
KENT
Scale: 1" = 4350'
LEGEND
Watershed Boundary
Subbasin Boundary
Stream Inflow Sampling Station

Figure 1. — Lake Waramaug watershed.

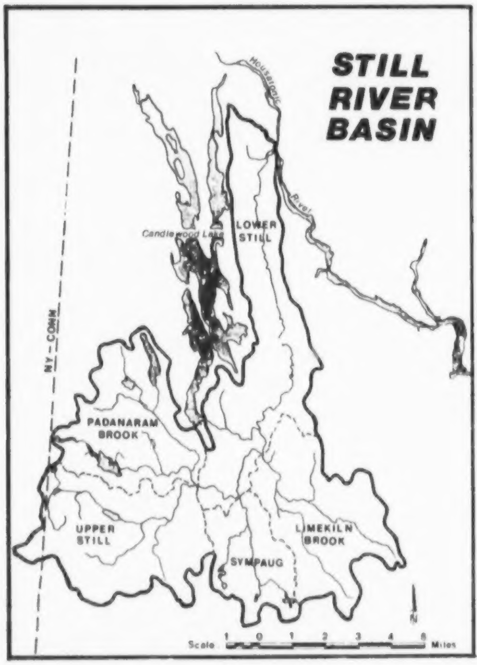
STILL RIVER BASIN
HOUSATONIC
River
Candlewood Lake
LOWER STILL
NY - CONN
PADANARAM BROOK
UPPER STILL
SYMPAUG
LIMEKILN BROOK
N
Scale 1 0 1 2 3 4 5 Miles

Figure 2. — Still River watershed.

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WHATEVER BECAME OF SHAGAWA LAKE?

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ABSTRACT

The response of Shagawa Lake, Minn. to an 80 percent reduction in external phosphorus loading, initiated in 1973 when a tertiary waste treatment plant began operation, is summarized. Total phosphorus concentration of the treatment plant effluent was 50 μg/l from 1973-1977. In November 1977 the level was raised to 400 μg/l . This change produced a 300 μg/l yr increase in total phosphorus loading to the lake from the treatment plant. Wastewater loading of total phosphorus accounts for 20 to 30 percent of the total external loading to the lake, decreased from 80 percent during pre-treatment years. Through 1978 the post-treatment response of the lake was stable with total phosphorus concentrations (average whole lake) reduced to about 60 percent of the pre-treatment levels (a reduction of ~ 40 percent), chlorophyll *a* (epilimnion) only slightly reduced, and Secchi disk depth increased slightly. The internal phosphorus loading phenomenon which has prevented complete recovery of the lake might be diminishing. Relationships between chlorophyll *a* and total phosphorus and between Secchi disk depth and chlorophyll *a* in Shagawa Lake are consistent with those established for other lakes.

INTRODUCTION

The response of Shagawa Lake, Minn. to a significant reduction in external phosphorus (P) loading provides a clear example of how internal P supplies can dely the recovery of heavily eutrophied lakes. Examples exist to document the predictable recovery of lakes when external P is reduced, based on predictions of P washout models of the type described by Sonzogni, et al. (1976), for example, Edmondson (1977) for Lake Washington, and Dillon, et al. (1978) for Gravenhurst Bay, Ontario. However, Shagawa Lake's response provides a case study in which these simple models are not adequate unless they are modified to account for internal loading.

Lake Norrviken, Sweden (Ahlgren, 1977), provides an example of an intermediate response — internal loading is important but baseline P levels decreased in accordance with washout model projections. The whole lake fertilization experiments which Schindler and co-workers (1974, and Schindler and Fee, 1975) conducted show no evidence of significant internal loading so if fertilization were halted, these lakes would be expected to respond as predicted by P washout models. Thus, the study of the long-term response of Shagawa Lake provides insight into the characteristic response of a lake toward one end of a continuum, the other end being demonstrated by lakes that respond as predicted by P washout models.

This paper will: (1) Extend observations covering Shagawa's response through 1976; and (2) compare the relationships between chlorophyll *a*, total phosphorus (TP), and Secchi disk depth in Shagawa Lake with those established for other relevant lake studies.

BACKGROUND

Bradbury (1978) described the development of European settlements around Shagawa Lake beginning in the late 1800's with the advent of mining and lumbering in this region of northeastern Minnesota. He showed how the sediments of the lake recorded this development through changes in chemical characteristics (especially P and iron), pollen types, and diatom and cladoceran remains. These changes document the lake's transformation from a relatively unproductive system to one of high productivity as population pressure increased in the watershed. Malueg, et al. (1975) demonstrated that wastewater phosphorus loading from the nearby community of Ely contributed 80 percent of the external TP loading in the late 1960's and early 1970's. Others have documented the high levels of algal productivity and biomass in the lake resulting from wastewater enrichment (Megard and Smith, 1974; Larsen, et al. 1975).

In 1973, a treatment plant which eliminated essentially all the wastewater P flowing to the lake began operating in Ely. The plant reduced the total external input from 6,200 to 7,200 kg/yr to 900 to 1,500 kg/yr, sufficient to reduce the average influent P concentration from 60 - 100 μg/liter to less than 20 μg/l. Since Shagawa's water retention time is short (less than 1 year) and the pre-treatment P retention time is even shorter (½ year), the lake could be expected to respond rapidly to reduced external P loading.

Larsen, et al. (1979) documented that the lake did indeed respond rapidly but this response was tempered by a resurgence of P from the sediments, especially during July and August. When this resurgence pattern was incorporated into a P mass balance model, the model tracked the temporal pattern well and mimicked the average values closely, but the average TP levels in the lake were well above those predicted from a simple TP washout model. One conclusion based on these simulations was that the TP resurgence had not diminished (through 1976) since the wastewaster treatment process began. About 2,000 to 2,500 kg of TP are released during the 2-month July through August interval.

METHODS

The sampling methods used for the lake and tributaries through September 1978 and the analytical procedures have been described elsewhere (Malueg, et al. 1975). From October 1976 to October 1978, the U.S. Forest Service at Ely collected and analyzed the samples under the direction of the U.S. Environmental Protection Agency. Only the central station, Brisson's Point South was sampled routinely. Samples were taken bi-weekly through June, weekly during July and August, then bi-weekly. All averages were time weighted. During 1979, there was no routine sampling program; however, Secchi disk data were obtained from S. Kliest (pers. comm. Vermilion Community College). During 1980, a limited sampling program (conducted by Vermilion Community College) included approximately weekly Secchi disk measurements at three stations and collection of an integrated water sample using a 5 m length of tygon tubing. Water samples were preserved with $HgCl_2$ (40 mg/l final concentration) and sent to the EPA's laboratory in Corvallis, Ore. for TP analysis. These observations cover the period from mid-May through the end of August.

For the Secchi disk, chlorophyll *a*, and TP regressions, we used data from the upper 5 m at Brisson's Point South for the period May-September, because this station provides the longest continuous record of values for these variables. For consistency with other studies, we chose the May-September interval as representative of summer conditions (e.g., Dillon and Rigler, 1974; Dillon, et al. 1978; Ahlgren, 1980).

Routine weekly tributary sampling ended in September 1978, so from that time, natural loadings were estimated by multiplying approximate total inflow by the average inflow TP concentration of all natural sources for the years 1969-1978 (see Larsen, et al. 1979). The representative value is 15.6 μg = TP/l. Estimated inflow came from a regression of measured annual inflows (1969-1977) against measured precipitation at the Winton Power Dam weather station, 8 km east of Shagawa Lake. The regression equation is: $F = 1.39\ PR - 18.8$ ($r^2 = 0.77$), where F is flow (x 10^6 m^3/yr) and PR is precipitation (cm/yr). To obtain wastewater loadings, treatment plant operators measured plant flows daily and phosphorus concentrations approximately every other day; loadings were calculated from these values (Jackson and Lindroos, 1980).

RESULTS

1. Changes in TP Supply — During 1977, the external input of TP into Shagawa Lake remained similar to that for the post-treatment years, 1974-1976 (Table 1). In November 1977, the chemical treatment process for P removal within the plant was modified for economic reasons to produce an effluent concentration of 400 g/liter, an increase from 50 μg/l for the interval from April 1973 to November 1977. This change in treatment increased the wastewater TP input by about 300 kg/yr to 400 kg/yr. In 1978, wastewater supplied 330 kg TP through September (Table 1) and an additional 95 kg from October through December (Jackson, pers. comm.). During 1979, wastewater supplied 330 kg (Table 1). A small unmeasured amount of wastewater bypassed the plant during 1978-1979. In previous years, this bypass has contributed less than 20 kg/yr TP to Shagawa Lake, so we expect that bypass was not significant during 1978 and 1979.

Since the annual natural supply ranged from 850 to 1,760 kg/yr, wastewater now accounts for 20 to 30 percent of the total supply of P to the lake. From 1977-1978, average inflow TP concentrations to the lake incorporating all sources were similar to the values for the 1974-1976 interval. These results indicate that for the period 1977-1979, TP loading to Shagawa Lake was similar to that for the 1974-1976 interval. The increase in annual TP loading associated with the revised wastewater effluent standard is masked by the natural variation in TP loadings.

Table 1. — Total phosphorus supplies to Shagawa Lake, Minn., 1977-1979.

	Natural Inflow			Wastewater Inflow			Combined Inflow		
Year	Flow x 10^6 m^3/yr	Conc. (μg/l)	Supply (kg)	Flow x 10^6 m^3/yr	Conc. (μg/l)	Supply (kg)	Flow x 10^6 m^3/yr	Conc. (μg/l)	Supply (kg)
1969-1972	61.1-125.1	14.1-17.3	1,060-1,760	1.2-2.1	2,600-4,200	5,040-5,460	62.3-127.2	58-100	6,230-7,200
1973-1976	41.5-102.0	14.7-20.5	850-1,630	1.3-1.7	[*]29-80	[*]40-130	42.8-103.7	[*]15.3-20.7	[*]890-1,490
1977	87.7	13.4	1,180	1.5	70	110	89.2	14.4	1,290
1978 (through Sept.)	65.5	12.6	830	1.10	340	330	66.6	17.4	1,160
1979	[†]71.4	[†]15.6	1,110	1.2	290	350	72.6	20.0	1,460

[*] Excludes average values for 1973 when wastewater treatment plant was not in operation for full year.

[†] Estimated as described in text.

2. Changes in TP, chlorophyll *a*, Secchi disk depth — The stable summer pattern of lake TP which developed in response to wastewater P reduction (Larsen, et al. 1979) continued through September 1978, when routine sampling of the lake was terminated (Figure 1). The internal loading event, identified previously, continued through 1978 (Larsen, et al. 1979, 1980). During 1977, the average TP concentration of the lake increased by approximately 50 μg/l from late May through August, corresponding to an increase of 2,650 kg TP within the lake supplied from internal sources. Similar increases were seen in previous years. The 1977 pulse apparently occurred about 1 month earlier than usual.

In other years, P concentration began to increase throughout the lake in late June and early July, then declined in late August and early September. In 1977, P concentration began to increase in late May and early June and declined in August. During 1978, average TP increased by approximately 30 μg/l corresponding to a lakewide increase of 1,600 kg. This is substantially lower than that seen in previous years. The average TP concentration in Shagawa Lake during 1977 and 1978 has remained similar to that measured during 1974-1976, indicating a reduction of approximately 40 percent over the pre-treatment period. (Table 2). This response is also seen in the immediate reduction and subsequent stability in average epilimnetic TP at Brisson's Point South for the May-September interval (Figure 2).

During June-August 1980, TP concentrations increased in the upper 5 m from 15.6 μg/l in early June to about 45 μg/l in August, corresponding to an increase of 1,150 kg (Figure 3). This increase is not directly comparable with the whole-lake increases reported previously; however, its magnitude is similar to that in 1978. The magnitude of these pulses in 1978 and 1980 is lower than that seen previously and may signal a reduction in the internal loading in Shagawa Lake. However, during both years, temperature profiles indicated that the lake was not severely stratified; temperatures in July and August near the bottom were only slightly lower than surface values. Oxygen depletion occurred, but its extent over the profundal zone was probably minimal due to the vertical mixing inferred from temperature profiles. Thus meterological conditions may have modified the release pattern seen in these years, while the potential for release may not yet have diminished.

In Shagawa Lake, a summer algal bloom typically develops in late June or early July and terminates in late August or early September, although some variation occurs in the timing. In the years since treatment began, the duration of the bloom has been shorter. During 1977 and 1978, this general pattern continued to occur (Figure 4). In July and August, chlorophyll *a* was similar to that seen in 1974-1976, nearly reaching pre-treatment concentrations (Table 2). There has been a substantial decrease in chlorophyll *a* during post-treatment years, but the impact is not as significant or as stable as that seen in the post-treatment TP pattern (Figure 2).

Changes in mean Secchi disk depth for May and June, July and August, and May-September at Brisson's Point South are summarized in Table 3. Most evident is the increased clarity during May and June after treatment began, corresponding to the associated reduction in algal biomass. Based on Secchi depth, the mean transparency increased by 0.8 m as compared to the pre-treatment period. When the 1978 value, which is similar to pretreatment values, is excluded, clarity increased 1 m in May and June. Later (July and August) no post-treatment improvement in Secchi depth is seen because algal biomass was not reduced significantly during this time period. Over the May-September interval, transparency increased by 0.5 m (0.7 m if the 1978 value is excluded) (Table 3 and Figure 2).

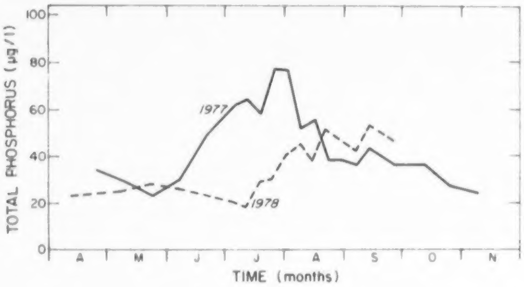

Figure 1. — Average total phosphorus concentrations in Shagawa Lake, 1977 and 1978, for the ice-free season. The values are whole lake, volume weighted.

Table 2. — Summary of changes in total phosphorus (whole lake) and chlorophyll *a* (upper 5 m) in Shagawa Lake, Minn. Numbers in parentheses are the ratios of the average concentrations of any 1 year to the mean of the 1971 and 1972 average concentrations.

	Total Phosphorus (μg/l)			Chlorophyll *a* (μg/l)	
Year	Annual Average	Ice-Covered Interval	Ice-Free Interval	May-June	July-August
*1971-1973	47.4-54.1	36.5-51.4	50.8-60.9	14.1-16.1	23.0-32.9
*1974-1976	29.3-31.4	19.4-24.6	34.6-35.7	6.4-11.4	15.5-33.4
1977	33.6(0.66)	23.8(0.60)	38.8(0.67)	7.2(0.46)	20.7(0.74)
1978 (through Sept.)	28.3(0.56)	19.6(0.49)	33.4(0.58)	14.0(0.90)	26.0(0.93)

*Values are ranges over the years shown from Larsen, et al. (1979).

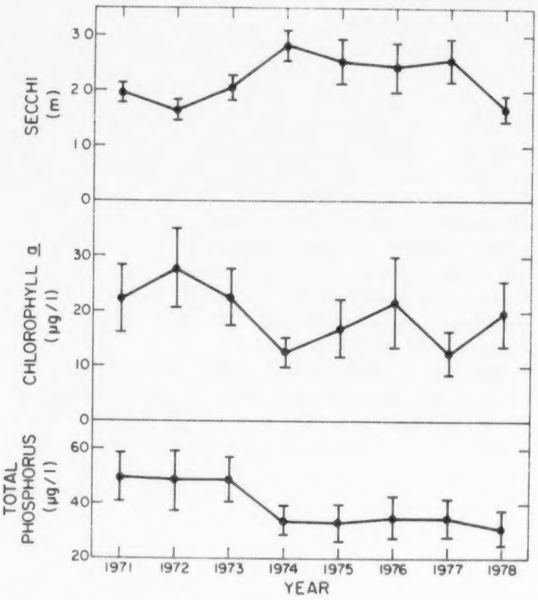

Figure 2. — Average summer (mid-way to mid-September) total phosphorus, chlorophyll *a*, and reciprocal Secchi disk depth at the central station, Brisson's Point South. Values are water column averages over the top 5 m.

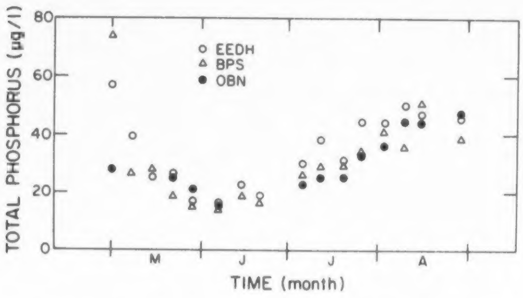

Figure 3. — Total phosphorus concentrations at three stations in Shagawa Lake, 1980. Values are for the upper 5 m.

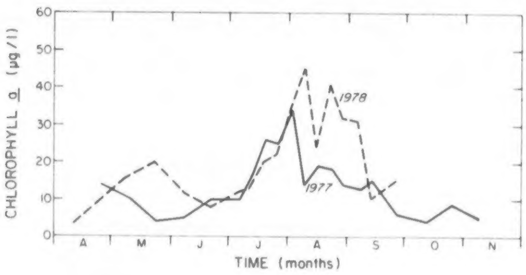

Figure 4. — Average chlorophyll *a* concentrations in Shagawa Lake, 1977 and 1978, for the ice-free season. The values are for the epilimnion (top 5 me), volume weighted.

Figure 5. — Chlorophyll *a* — (Figure 4b) relationships for the Shagawa Lake — Burntside Lake combination. Darkened circles are Burntside Lake, open circles are Shagawa Lake.

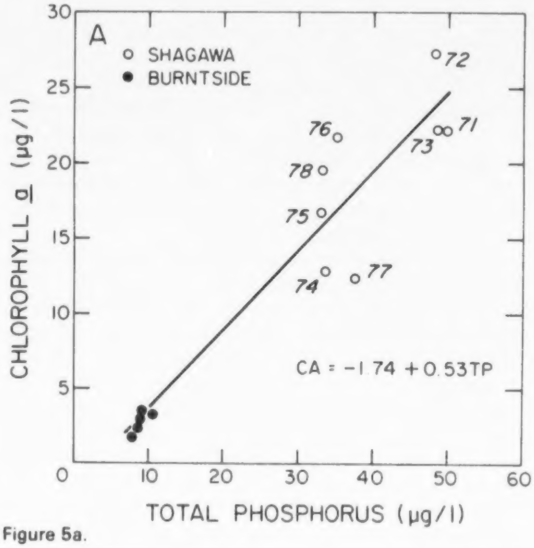

Figure 5a.

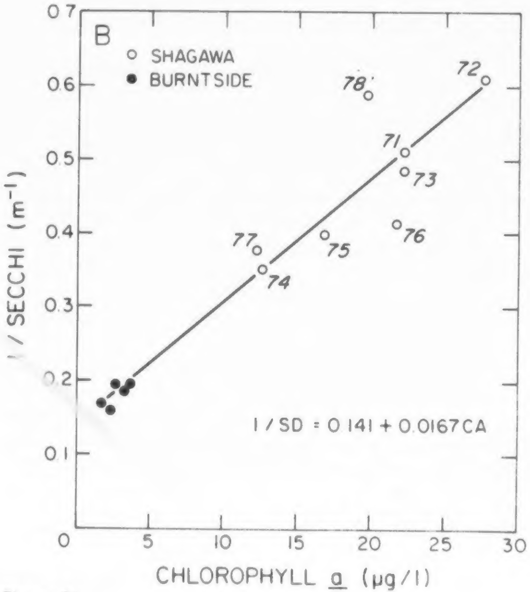

Figure 5b.

RELATIONSHIPS AMONG SD, CA, AND TP

Figure 5 summarizes chlorophyll *a*, TP, and Secchi depth - chlorophyll *a* relationships for the Shagawa Lake - Burntside Lake combination. Burntside Lake lies 10 river km upstream of Shagawa Lake. Its outlet, the Burntside River, accounts for 60 to 70 percent of Shagawa's water inflow. We chose to include the Burntside Lake TP, chlorophyll *a*, and Secchi depth values because we believe that further declines in Shagawa's TP concentration will be accompanied by changes in chlorophyll *a* and Secchi depth which follow the regression lines. The regressions display significant linear relationships among the variables (Table 4, ~ 0.05). The chlorophyll *a* - TP relationship suggests that 1 μg of TP produces 0.5 μg of chlorophyll *a*, regardless of whether Burntside Lake values are included. The intercepts do not differ from zero. The Secchi depth - chlorophyll *a* relationship for the Shagawa - Burntside Lake combination implies a background SD of 7.1 m. For Shagawa Lake alone, the value is slightly lower and is not statistically different from zero ($\alpha = 0.05$).

Reciprocal Secchi depth is chosen as an independent variable to be consistent with the arguments developed in Lorenzen (1980) and Megard, et al. (1980). Reciprocal Secchi depth can be expected to change as a linear function of chlorophyll *a* as:

$$1/\mathrm{SD} = \alpha + \beta\,\mathrm{CA}$$

where α is the reciprocal of background Secchi depth and β is related to the partial extinction of light by chlorophyll *a*: $\beta = K_c/\ln(I_o/I_z)$ where I_o and I_z are surface irradiance and irradiance at the Secchi disk depth, and K_c is the partial attenuation of irradiance due to chlorophyll *a*.

DISCUSSION

Post-treatment chlorophyll *a* and TP data obtained since 1976 in Shagawa Lake are consistent with those obtained for the years immediately after P loading was reduced. The post-treatment TP data show that Shagawa Lake reached an equilibrium in response to reduced P loading rapidly and that the lake has remained at this level through 1978, as indicated by the low year to year differences in mean (Figure 2). The responses of chlorophyll *a* and Secchi depth have not been as clear, although post-treatment values are lower than those for pre-treatment over the same intervals (Tables 2 and 3).

Average TP in Shagawa declined in response to loading reduction, but not to the extent predicted from P washout models because internal sources have supplied significant amounts of P (Larsen, et al. 1979, 1980). This internal source is probably the sediments of the profundal plain and deep-hole areas of the lake which release P after anaerobic conditions have developed (Armstrong and Stauffer, 1980). The magnitude of this internal supply does not appear to have diminished over the post-treatment period through 1977. This post-treatment pattern contrasts with that seen in Norrviken in which internal P loading declined significantly through time (Ahlgren, 1977). The data obtained during 1978 and 1980 indicate that the magnitude of internal loading may be declining.

Although Shagawa Lake has not responded according to projections from TP washout models, relationships among Secchi depth, chlorophyll *a*, and TP are consistent with those seen for other lakes which have shown significant recoveries after extended TP loading reduction, as demonstrated by data from Lakes Washington and Norrviken, and Gravenhurst Bay. Table 5 summarizes the relationships for these studies.

These cases were selected for comparison with Shagawa Lake because P loading was reduced sharply and because many years' data are available which document the pre-treatment conditions and the recovery patterns for each lake. The data for Washington were obtained from Edmondson (1977) and Smith and Shapiro (1980), for Gravenhurst Bay from Dillon, et al. (1978), and for Norrviken from Ahlgren (1980, and pers. comm.). The 1971 chlorophyll *a* - TP data pair for Norrviken was excluded from the chlorophyll *a* - TP regression because chlorophyll *a* is an obvious outlier (see Ahlgren, 1978).

The relationships summarized in Table 4 for Shagawa Lake and Table 5 for Lakes Washington and Norrviken and Gravenhurst Bay indicate that Shagawa's response falls within the range of values characteristic of these relationships for the other cases. The slope of the chlorophyll *a* - TP relationship for Shagawa Lake falls between that for Lakes Washington and Norrviken while the slope of the Secchi disk depth - chlorophyll *a* relationship is lower than those for Lake Washington and Gravenhurst Bay but higher than that for Norrviken. This indicates that although the Shagawa Lake TP response to loading reduction was unique, the control exhibited by TP on chlorophyll *a* on Secchi depth is similar to that for other lakes and that further declines in lake TP can be expected to produce further declines in chlorophyll *a* and increases in transparency consistent with that seen at other sites.

In summary, Shagawa Lake continues to display a stable pattern in total phosphorus concentration which was reached rapidly after external phosphorus input was reduced by wastewater treatment. The average total phosphorus concentration and seasonal patterns continue to be controlled by internal loading during summer months. Chlorophyll *a* and Secchi disk transparency have changed only slightly since treatment began. We now know that, in Shagawa Lake, chlorophyll *a* responds to total phosphorus and Secchi disk depth to chlorophyll *a* in a manner similar to that seen for other lakes. Thus, further changes in algal biomass and transparency are expected only if total phosphorus declines further.

Table 3. — Summary of Secchi disk depth (meters) in Shagawa Lake, Minn. for the period 1971-1980 at Brisson's Point South. Numbers in parentheses are the ratios of the average values for a particular year to the mean of the 1971-1972 average values..

	May-June	July-August	May-September
1971	2.08 (1.02)	2.13 (1.20)	1.95 (1.09)
1972	1.98 (0.98)	1.44 (0.81)	1.63 (0.91)
1973	2.28 (1.12)	1.97 (1.11)	2.05 (1.14)
1971-1973 mean	2.11 (1.04)	1.85 (1.04)	1.88 (1.05)
1974	2.74 (1.35)	2.70 (1.52)	2.81 (1.57)
1975	3.19 (1.57)	1.77 (1.10)	2.50 (1.40)
1976	3.31 (1.63)	1.71 (0.96)	2.42 (1.35)
1977	3.24 (1.60)	2.09 (1.17)	2.63 (1.45)
1978	2.07 (1.02)	1.36 (0.76)	1.68 (0.93)
1979	————	1.61 (0.91)	————
1980	2.94 (1.45)	1.62 (0.91)	————
1974-1980 mean	2.92 (1.44)	1.88 (1.05)	2.40 (1.34)

Table 4. — Relationships among Secchi disk depth **chlorophyll *a*** and total phosphorus in Shagawa and Burntside Lakes, Minn. Slopes and intercepts are given with 95 percent confidence limits. r^2 is the linear correlation coefficient, and n is the number of data pairs.

Chlorophyll-*a* vs TP				
Shagawa - Burntside	-1.74 ± 3.58	0.529 ± 0.121	13	0.89
Shagawa	1.17 ± 10.12	0.460 ± 0.100	8	0.44
Secchi disk depth vs **chlorophyll *a***				
Shagawa — Burntside	0.141 ± 0.035	0.0167 ± 0.002	13	0.92
Shagawa	0.185 ± 0.223	0.0146 ± 0.011	8	0.63

Table 5. — Relationships among Secchi disk depth **chlorophyll *a*** and total phosphorus in Lakes Washington and Norrviken and in Gravenhurst Bay. Slopes and intercepts are given with 95 percent confidence limits. r^2 is the linear correlation coefficient, and n is the number of data pairs.

Chlorophyll *a* vs TP				
Gravenhurst Bay	-1.61 ± 6.98	0.267 ± 0.141	7	0.72
Norrviken	13.30 ± 22.60	0.406 ± 0.138	10	0.85
Washington	-4.37 ± 3.92	0.597 ± 0.110	15	0.91
Secchi disk depth vs **chlorophyll *a***				
Gravenhurst	0.187 ± 0.137	0.0217 ± 0.016	7	0.71
Norrviken	0.505 ± 0.321	0.0107 ± 0.004	10	0.79
Washington	0.204 ± 0.073	0.0235 ± 0.004	18	0.91

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A RETROSPECTIVE LOOK AT THE EFFECTS OF PHOSPHORUS REMOVAL IN LAKES

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ABSTRACT

A retrospective look at 16 north temperate lakes which have undergone restoration shows that the reductions in chlorophyll *a* which accompanied phosphorus removal were typically immediate and continuous. However, the exact response of each lake to P removal was unique. It is suggested that these differences in response result to a large extent from changes in TN:TP which accompany restoration. A variable chlorophyll yield model, which depends explicitly on the TN:TP ratio, is presented and tested using data from Lake Norrviken (Sweden). The new model appears to greatly reduce chlorophyll prediction error in lakes which are undergoing restoration.

INTRODUCTION

The prediction of algal biomass in lakes undergoing changes in nutrient loading is a topic of great concern for lake management, and during the last decade a number of eutrophication models have been developed for this purpose. In this regard, the Dillon-Rigler (1975) and Vollenweider (1976) models have been widely used to justify phosphorus control as a lake restoration measure. However, with notable exceptions (e.g. Dunst, et al., 1974; Ryding and Forsberg, 1975; Born, 1979), few systematic evaluations have been made of the response of a large number of lakes to restoration measures. With this in mind, Smith and Shapiro (1980) have analyzed the response of algal biomass to successful phosphorus reduction in 16 lakes, using data from the literature. The purpose of this paper is to summarize the important features of that analysis, and to present a model which helps explain a major portion of the variance associated with chlorophyll-phosphorus regressions noted in that and other studies.

VARIABILITY IN THE RESPONSE OF LAKES TO PHOSPHORUS REMOVAL

We have analyzed the changes in algal biomass in 16 north temperate lakes where nutrient abatement, or natural variation in phosphorus loading, has led to measurable reductions in concentrations of total phosphorus in the lake. Growing season mean values of chlorophyll *a* (c), total P (TP), and total N (TN) for these lakes are summarized in Smith and Shapiro (1980).

When the data were examined using standard regression and correlation techniques (Steel and Torrie, 1960), significant regressions between (c) and TP were found for nine of the 16 lakes. The chlorophyll-phosphorus relationships for two of these lakes are shown in Figure 1, in which a significant regression is evident for Norrviken (Figure 1b), but not for Oxundasjon (Figure 1a). An important aspect of the analysis thus is that the response of individual lakes to a reduction in TP is unique. Some lakes do show a good relationship and some do not. Even among those that do, however, a statistical comparison of their chlorophyll-phosphorus relationships shows that significant differences ($P < 0.05$) exist between the slopes and intercepts of their regressions (Table 1). The variability in response of these lakes is shown in Figure 2, in which the cloud of data is compared to the nine individual regression lines.

Two important points emerge from Figure 2. First, it is clear that the difference in response of individual lakes to changes in total P can account for a major proportion of the variance commonly noted in "global" chlorophyll-phosphorus regressions (e.g. Dillon and Rigler, 1974; Jones and Bachman, 1976; Nicholls and Dillon, 1978). Second, the unique response of each lake raises questions regarding an assumption made by many users of current global eutrophication models (e.g. the models of Dillon and Rigler, 1975; Lee, et al. 1978; and Vollenweider, 1976) — the assumption that individual lakes will respond in a similar fashion to a given change in total phosphorus.

THE IMPORTANCE OF TN:TP RATIOS TO CHLOROPHYLL YIELD

If we are to develop eutrophication models that more accurately predict the response of lakes to changes in phosphorus loading, it is important that we understand the sources of variability which generate the scatter typically observed in chlorophyll-phosphorus regressions (e.g. Figure 2a). In their recent review Nicholls and Dillon (1978) discussed several reasons for the scatter, including methodological variation, the relative biological availability of phosphorus in different lake waters, and variations in the chlorophyll/algal cell volume ratio. However, these authors did not consider a major factor pointed out by Sakamoto (1966): The yield of chlorophyll at a given concentration of total P is sensitive to variations in the $\overline{TN}:\overline{TP}$ ratio. In his original analysis, Sakamoto considered nitrogen to limit algal biomass in lakes where $TN:TP < 10$; similarly, he considered phosphorus to be limiting where $\overline{TN}:\overline{TP} > 17$. He also felt that chlorophyll was proportional to either $\overline{TN}$ or $\overline{TP}$ in lakes where $10 < \overline{TN}:\overline{TP} < 17$. Thus, nitrogen availability should influence the chlorophyll response to phosphorus over a broad range of $\overline{TN}:\overline{TP}$ ratios. The importance of the N:P ratio has since been commented upon by Chiaudani and Vighi (1974), Porcella, et al. (1974), Schindler (1977), Allan and Kenney (1978), Forsberg, et al. (1978a) and Allan (1980).

This influence of the TN:TP ratio on chlorophyll yield during lake restoration can be seen in Norrviken and Oxundasjon (Sweden) (Figure 1). In the case of Norrviken (Figure 1b), diversion of wastewater led to consistent declines in both TP and c (Ahlgren, 1978, and 1980.) The only exception occurred in 1970, when the algal biomass appeared to be N-limited ($TN:TP = 8$), and may have been regulated by intense zooplankton grazing as well (Shapiro, 1979). In Oxundasjon, the effect of changes in nutrient limitation is also evident (Figure 1a): during the 6 years for which data are available, the $\overline{TN}:\overline{TP}$ indicated N-limitation in 1971 ($\overline{TN}:\overline{TP} = 6{:}4$), and either N- or P-limitation in other years ($\overline{TN}:\overline{TP} = 9.8 - 13.9$.). As would be predicted from Sakamoto's (1966) analysis, only when the data from 1971 are excluded is a marked relationship evident between $\overline{TN}$ and $\overline{c}$.

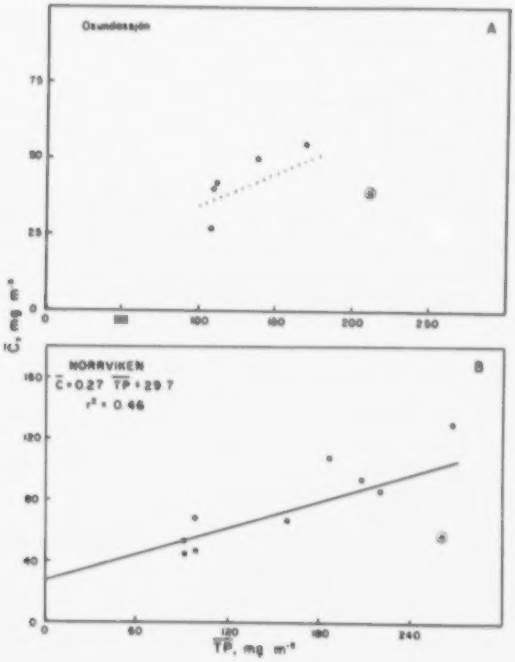

Figure 1. — Phosphorus dependence of chlorophyll *a* in (A) Lake Oxundassjön 1970-1975, and (B) Lake Norrviken 1969-1978. Confidence limits for the slope and intercept in (B) are m ± 0.44 and b ± 1.01. Circled points denote years of probable N-limitation ($TN:TP < 10$). Modified from Smith and Shapiro (1980).

Table 1. — A. Lower left — logarithmic regressions; upper right — arithmetic regressions. Letter designates significant ($P \leq 0.05$) difference between two slopes.

Lake	W	C	T	Gr	G	L	B	N	S
W		-	-	-	a	-	a	-	-
C	-		-	-	-	-	a	-	-
T	-	-		-	-	-	-	-	-
Gr	a	-	-		a	-	a	a	-
G	-	-	-	-		a	a	-	-
L	-	-	-	a	-		a	a	-
B	a	-	-	a	-	a		-	-
N	-	-	-	a	-	-	-		-
S		-	-	-	-	-	-	-	

Table 1. — B. Same format as above, except letter designates significant differences between two intercepts.

Lake	W	C	T	Gr	G	L	B	N	S
W		-	-	a	-	-	-	-	-
C	-		-	-	-	-	-	-	-
T	-	-		-	-	-	-	-	-
Gr	a	-	-		-	-	a	-	-
G	-	-	-	-		-	-	-	-
L	-	-	a	a	-		-	-	-
B	a	-	a	a	-	-		-	-
N	-	-	a	a	-	-	-		-
S		-	-	-	-	-	-	-	

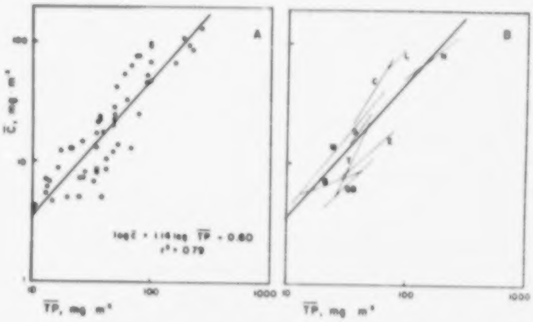

Figure 2. — (A) Phosphorus dependence of chlorophyll *a* in nine north temperate lakes in which $\overline{TN}:\overline{TP} < 10$. Each point represents a single growing season mean. (B) Same as (A), except each line represents the regression line for an individual lake. E-lake Ekoln; GB-Gravenhurst Bay; other symbols as in Table 1.

A VARIABLE YIELD CHLOROPHYLL-PHOSPHORUS MODEL

As can be seen in Table 2, various lake restoration measures lead not only to reductions in total P, but also lead to changes in the $\overline{TN}:\overline{TP}$ ratio. In fact, Table 2 suggests that the $\overline{TN}:\overline{TP}$ ratio typically increased in these lakes, regardless of the type of restoration

Table 2

Lake	Years	Range of $\overline{TN}:\overline{TP}$	Trend*	Restoration method	References
Washington	1957-1975	8.8-25.2	increase	wastewater diversion	W.T. Edmondson, pers. comm.
Norrviken	1969-1978	8.1-17.9	increase	wastewater diversion	I. Ahlgren, pers. comm.
Edssjön	1970-1975	7.5-11.3	increase	wastewater diversion	I. Ahlgren, pers. comm.
Oxundasjön	1970-1975	6.4-13.9	increase	wastewater diversion	I. Ahlgren, pers. comm.
Gravenhurst Bay	1969-1975	10.7-28.9	increase	wastewater P removal	Dillon, et al. 1978
Ekoln	1972-1975	14.6-71.8	increase	wastewater P removal	Forsberg, et al. 1978b
Boren	1973-1976	10.2-38.9	increase	wastewater P removal	Forsberg, et al. 1978b
Ramsjön	1972-1974	3.6-7.3	increase	wastewater P removal	Ryding and Forsberg, 1975
Ryssbysjön	1973-1974	3.8-5.2	increase	wastewater P removal	Ryding and Forsberg, 1975
Cline's Pond	1970-1971	9.0-18.5	increase	nutrient precipitation	Funk and Gibbons, 1979

method used. It thus appears that variations in total nitrogen, as well as changes in total phosphorus, must now be considered in restoration efforts. A mechanistic, variable chlorophyll yield model which explicitly considers variations in the ($\overline{TN}:\overline{TP}$ ratio has been developed for this purpose by Smith (1980). In general, the model predicts a family of parallel chlorophyll-phosphorus curves (Figure 3) described by the following:

$$\log \bar{c} = 1.55 \log \overline{TP} - b,$$

where the y-intercept, b, is a function of the $\overline{TN}:\overline{TP}$ ratio:

$$b = 1.55 \log \left[\frac{6.404}{0.0204(\overline{TN}:\overline{TP}) + 0.334} \right]$$

The derivation and assumptions of this variable yield model are discussed by Smith (1980).

One feature of the model which is evident in Figure 3 is that many trajectories of change in chlorophyll *a* are possible for a given reduction in total phosphorus. For example, it appears from Figure 3 that chlorophyll may actually *increase* with a reduction in $\overline{TP}$ if the $\overline{TN}:\overline{TP}$ ratio increases sufficiently during restoration. Such a trend was actually observed in Oxundasjon between the years 1971 and 1972 (Figure 1a) and in Norrviken between 1970 and 1971 (Figure 1b). Furthermore, Figure 3 shows that a marked reduction in total P may also lead to no change in chlorophyll *a* if there is a modest change in $\overline{TN}:\overline{TP}$. This pattern was observed in Lake Norrviken (1974-75), when TP dropped from 158 to 98 mg m^{-3}, but TN:TP rose from 12.3 to 17.9. As a result, the concentration of chlorophyll *a* remained essentially constant (67 to 68 mg m^{-3}).

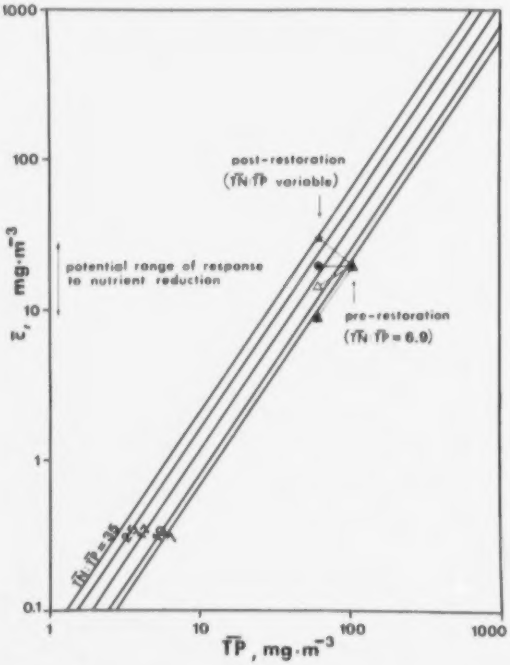

Figure 3. — Graphical display of the variable yield model, showing four potential trajectories of change in chlorophyll *a* following reduction of TP from 100 to 60 mg m

The variable yield model thus makes general predictions which appear to be confirmed in actual restoration experiences. However, a detailed comparison of the variable yield model with the Dillon-Rigler (1974) model emphasizes the greater accuracy of chlorophyll prediction when $\overline{TN}:\overline{TP}$ ratios are considered. An analysis of 20 lakes for which $\overline{TN}:\overline{TP}$, $\overline{TP}$, and c were known was made (Smith 1980), in which the changes in chlorophyll *a* were predicted using the Dillon-Rigler (1974) model and the variable yield model. The predictions (c_{pred}), which were based on observed changes in TP during restoration, were then compared to the actual changes in chlorophyll which occurred in the lakes (c_{obs}). The results of the analysis for Norrviken (Figure 4) are typical of the pattern noted for the remaining lakes. With the exception of the last 4 years of restoration, the Dillon-Rigler model consistently overestimates the concentrations of chlorophyll actually observed in Norrviken (Figure 4a). The variable yield model, however, much more closely predicts the changes in c (Figure 4b). The improved accuracy of the variable yield model is clearly shown by a 90 percent reduction in the total prediction error, estimated here as the sum of squares (SS):

$$SS = (c_{pred} - c_{obs})^2.$$

When all the values of chlorophyll *a* predicted from Equation 1 and 2 are regressed on the measured values of $\overline{TP}$ for Norrviken, the slopes and intercepts of the regressions:

$$c_{pred} = 0.570\,\overline{TP} - 24.3,\ r^2 = 0.85$$
$$m \pm 0.195,\ b \pm 35.3$$

$$\log c_{pred} = 1.277\,\overline{TP} - 1.009,\ r^2 = 0.92$$
$$m \pm 0.297,\ b \pm 0.653$$

are not significantly different from those actually observed (Smith and Shapiro, 1980) (cf. Figure 1b).

It should be pointed out that this comparison considers 3 years (1970-1972) during which the $\overline{TN}:\overline{TP}$ ratio was 12. Dillon and Rigler (1974) point out, however, that their model should not be used in such cases. Nonetheless, even when these years are excluded, the variable yield model generates 75 percent less prediction error (SS = 4169) than does the Dillon-Rigler model (SS = 15551).

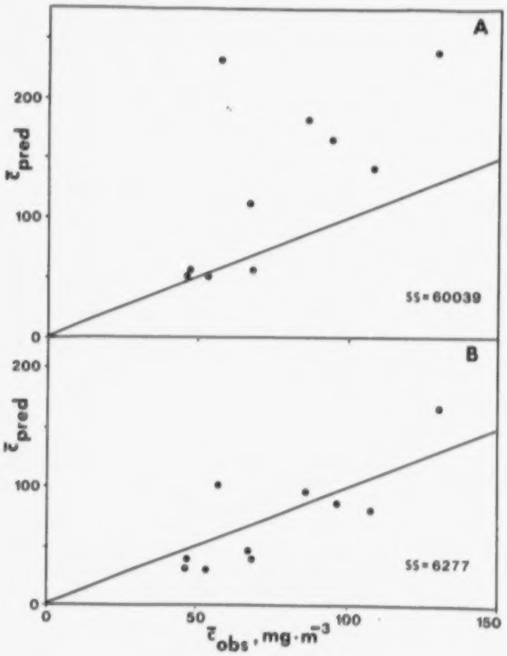

Figure 4. — Comparison of observed concentrations of chlorophyll *a* (c_{obs}) in Lake Norrviken with (A) the predictions made by the Dillon and Rigler (1974) model, and (B) the predictions made by the variable yield model. (SS = sum of squared deviations of predicted values from observed concentrations of chlorophyll; see text.)

CONCLUSIONS

A retrospective look at 16 north temperate lakes which have undergone restoration has provided evidence that the reductions in chlorophyll *a* which accompanied successful phosphorus reduction were almost always immediate and continuous (Smith and Shapiro, 1980). However, the individual lakes behaved uniquely in their response to nutrient removal. The $\overline{TN}:\overline{TP}$ ratio also showed marked long-term changes in the cases where nitrogen data were available, as well. We believe that these changes in $\overline{TN}:\overline{TP}$ modified the quantitative response of the algae to the declines in total P, and were, to a large extent, responsible for the significant differences noted in the slopes and intercepts of the chlorophyll-phosphorus regressions for the individual lakes (Table 1; Figure 2b).

Because the TN:TP ratio typically increases over the course of restoration (Table 2) the chlorophyll yield per unit total P in restored lakes can also be expected to increase and may tend to offset the potential benefits of phosphorus removal. Although the majority of lakes do appear to be P-limited on the basis of the $\overline{TN}:\overline{TP}$ ratio (Jones and Bachmann, 1978; Weiss, 1979; Smith, unpubl.), lakes having a low $\overline{TN}:\overline{TP}$ are not uncommon in many regions (e.g. Florida, R. E. Carlson, pers. commun.; Denmark, Lastein and Gargas, 1978; Sweden, Ahlgren, 1980). Management strategies in these regions should take this fact into account. We believe, with Dillon and Rigler (1974), and Allen (1980), that the use of current eutrophication models is inappropriate for these lakes, and we hope that the model presented here and in Smith (1980) will help predict conditions in such lakes following restoration.

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SIGNIFICANCE OF SEDIMENTS IN LAKE NUTRIENT BALANCE

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ABSTRACT

A considerable part of phosphate entering a lake will enter the sediment; the concentration of the phosphate in the lake will therefore be lower than when calculated as a conservative compound. It has been suggested that the amount in sediments is a proportion of the phosphate entering the lake, or that the phosphate in sediments is a constant fraction of the concentration. Mathematically, it can be shown that for lakes in a steady state these assumptions are identical. Recently it has been shown that better results could be obtained for some lakes on the assumption that the amount which is in sediments is controlled by adsorption on the sediments; adsorption isotherm can be used to describe this process. If this is the case, the amounts of phosphate in the sediments are controlled by the phosphate concentration in the lake and by the total sediment load of the lake. Variation in the sediment load can probably explain a large part of the scatter in statistical (stochastic) phosphate models. There is some indication that due to sedimentation the water retention time controls the amount of phosphate which is retained in the lake. It seems likely that phosphate profiles in lake sediments can give semi-quantitative rapid information of the loading history.

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PREDICTING DREDGING DEPTHS TO MINIMIZE INTERNAL NUTRIENT RECYCLING IN SHALLOW LAKES

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ABSTRACT

In shallow eutrophic lakes alternating periods of temperature stratification and wind-induced turnover events can produce a discontinuous but significant flow of phosphorus released from the sediments to the photic zone. The result can be sequences of weather-dependent algal blooms. The phosphorus release is typically associated with oxygen depletion of the water near the bed. The mixing events are caused by strong winds. A method has been developed to predict vertical temperature structures and multiple turnover events in shallow lakes in response to wind forces and heat transfer from the atmosphere. The method is an extension of the Minnesota Lake Temperature Model and based on integral energy transfer. It has been verified against field measurements of stratification structures for a time scale of 12 hours. With several years of recoded weather data as input, the sequence and number of turnover events in the Fairmont Lakes in southern Minnesota have been determined for selective alternative dredging depths. It was possible to determine a relationship between the number of midsummer turnover events, the number of stratification periods of 5 days or more, and the dredged lake depth. It was therefore possible to estimate which dredged conditions would reduce summer fertilization of the photic zone by phosphorus recycled from the lake bed.

INTRODUCTION

Dredging is one of several methods to restore shallow and eutrophic lakes. Some dredging techniques and case studies are summarized in Dunst, et al. 1974, and U.S. EPA, 1979. This paper describes a method to determine the depth to which a lake may be dredged to prevent phosphorus recirculation from the sediments. The method of computation is for shallow lakes. It will be illustrated by using the Fairmont Lakes in southern Minnesota.

CONCEPT

It has been found (Stefan and Hanson, 1979, 1980) that in very shallow, eutrophic lakes much of the phosphorus necessary for the growth of phytoplankton, including nuisance blooms of blue-green algae in the summer, can be recycled from the bottom sediments. Several mechanisms will release phosphorus from the sediments to the water: (a) Chemical release when the hypolimnetic waters become anaerobic during stratification; (b) uptake by the roots of macrophytes and release through remineralization; and (c) release through the digestive tract of bottom feeders. It is therefore not always true that only phosphorus loading from runoff produces nuisance blooms in shallow lakes.

Dredging of a shallow lake usually does not remove all phosphorus-containing materials from a lake bed. Often newer layers of deposit are removed, exposing older layers. If the benthic material is organic, phosphorus will still be present in the sediments after dredging. The release processes of phosphorus also will not be significantly altered by dredging. The success of a dredging program must therefore not be related to the availability of phosphorus as a nutrient but to other factors:

1. Dredging changes summer stratification and vertical mixing characteristics by increasing depth. This is illustrated in Figure 1, which displays the simulated summer isotherms of the same lake under the same weather conditions for three different depths. Deepening the mixed layer or complete overturns brings the phosphorus released on the lake bottom to the photic zone near the lake surface, where it can be used by phytoplankton. Greater depth reduces the frequency of summer overturns in very shallow lakes.
2. The greater depth provides a larger volume of hypolimnetic water which in turn contains a larger quantity of oxygen. Given identical rates of benthic oxygen uptake per unit area, the hypolimnion of a deeper lake will take longer to become anaerobic than the hypolimnion of a more shallow lake. Phosphorus release thus will be delayed in the deeper lake.
3. A third and minor effect of dredging is to reduce water temperature by increasing lake volume. The water temperature depression increases oxygen solubility and decreases biological kinetic rates; it thereby delays oxygen depletion in the hypolimnion and slows growth rates of algae.

The general concept is that shallow eutrophic lakes can be dredged to such a depth that phosphorus released from the sediments into the hypolimnion is not recycled to the photic zone by lake overturns. This will reduce the standing crop of algae. A method to determine the required dredging depth will be presented and illustrated.

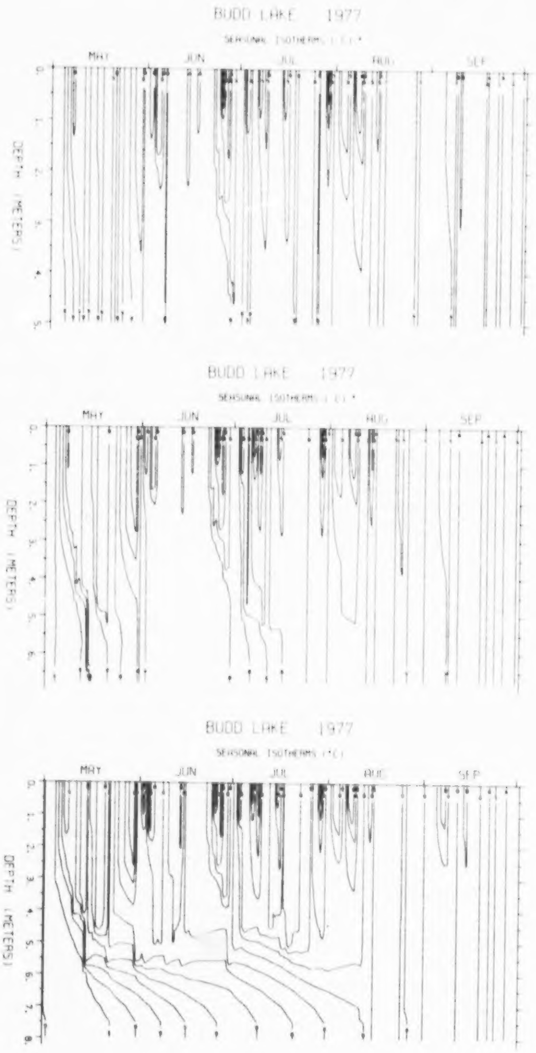

Figure 1. — Simulated isotherms, Budd Lake, 1977.

FAIRMONT LAKES STRATIFICATION, PHOSPHORUS RECYCLING, AND MIXING STUDY

Problem Description

The City of Fairmont in southern Minnesota has used several different strategies to reduce algae blooms in its chain of five very shallow city lakes. Treatment with copper sulfate as well as diversion and treatment of municipal sewage effluent have not solved the problem permanently. Since 1966, the city has been pursuing a dredging program.

The original basins of the Fairmont Lakes were formed by melting of ice blocks in the postglacial period. They have been filled with as much as 12 to 15 meters of lake-derived organic materials. Area versus depth curves for undredged and anticipated dredged conditions of one of the lakes are shown in Figure 2. Conditions for the other lakes are similar.

The lakes have surface areas ranging from 0.34 to 2.25 km² and mean depths from 2.1 to 3.7 meters. Water budgets for the years 1973, 1974, and 1975 showed that hydraulic residence times varied with weather from 0.2 years to 3.1 years. Much of the runoff occurs during snowmelt.

Primary productivity in the shallow, eutrophic Fairmont Lakes appears alternately limited by light and by phosphorus availability. Phosphorus is the basic material prerequisite. Light availability is often the dynamic regulatory parameter, and is dependent on solar radiation intensity, light attenuation in the water, and mixed layer depth. Attenuation in turn depends on the color and the suspended material content of the water.

Phosphorus budgets for the Fairmont Lakes have been presented by Barr (1974), Knoll and Megard (1973), and Stefan and Hanson (1979, 1980). They offer strong evidence that phosphorus loading by surface runoff from rainfall or snowmelt or from municipal waste water cannot account for the total summer phosphorus used by the algae.

Observed phosphorus and chlorophyll *a* data measured in the summer of 1979 are shown in Figure 3. A line for phosphorus limitation has been added.

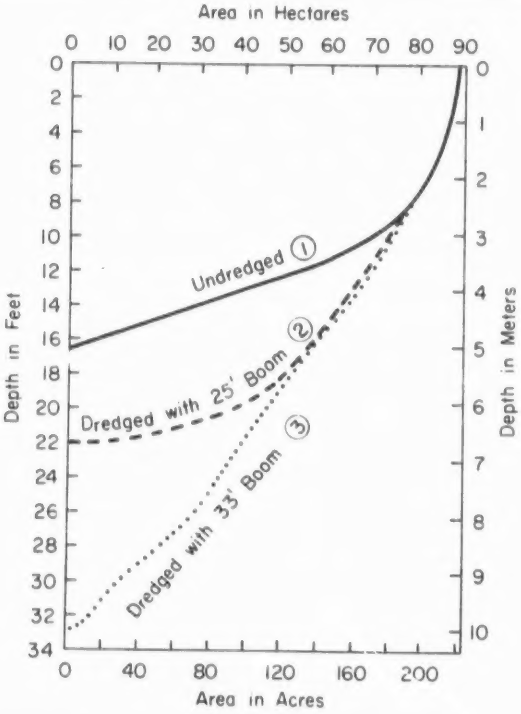

Figure 2. — Depth/area relationships for Budd Lake.

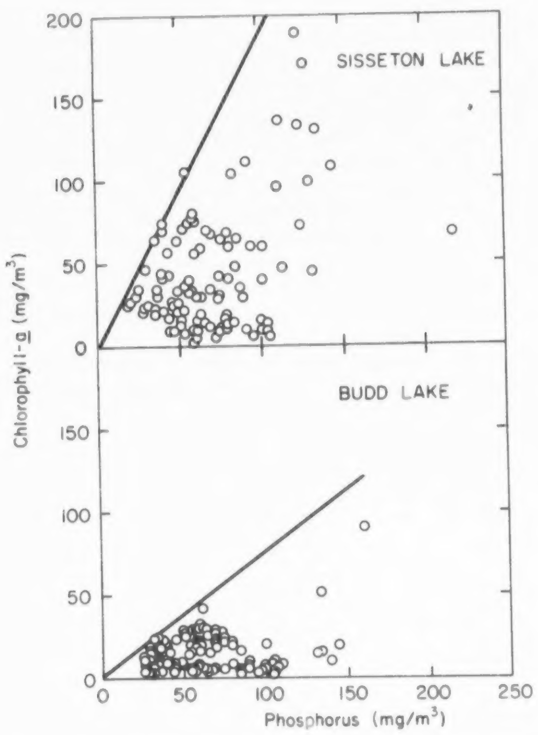

Figure 3. — Relationship between chlorophyll *a* and total phosphorus in 1979. (Budd Lake treated with copper sulfate.)

Computer Model Simulations of Stratification Structures and Vertical Mixing

The summer stratification dynamics and the effects of dredging on vertical mixing were simulated in a mathematical model.

The temperature stratification and the vertical mixing in a shallow and small lake (large lakes are usually not considered for dredging because of cost) are usually the result of intensive air-water interaction. Transfer of heat and wind energy is of great importance. Recent advances (Ford and Stefan, 1980; Harleman and Octavio, 1977; Stefan and Ford, 1975; Tucker and Green, 1977) in the quantitative description of these interactions make it possible to predict a lake's temperature structure with reasonable accuracy on a daily or even shorter time scale, provided that the forcing weather parameters are known.

In the Minnesota Lake Temperature Model (MLTM) (Ford and Stefan, 1980; Stefan and Ford, 1975), the lake is considered as a stack of horizontal layers of equal thickness z (e.g., 0.5 m) and variable horizontal area A(z). Each of the layers is considered to be homogeneously mixed and isothermal. A density difference between layers resulting from thermal differences restores horizontal stratification if the system is brought out of equilibrium. Water temperature is considered variable with depth and time. Temperatures T (i, k) for specific layers (i) at a particular time (k) are computed by the model. The discrete element approach gives the vertical temperature distribution in the form of a step function. The meteorological variables (air temperatures, dew point temperature, wind, direction, solar radiation, and wind speed) are input data.

It has been observed that the turbulence generated in a lake by wind or natural convection mixes homogeneously the upper layers to a depth called the mixed layer depth. The mixed layer depth can range from a few centimeters to the total depth of the lake. The mixed layer depth and the temperatures of the layers are found in the MLTM by applying internal (thermal) energy balances and mechanical energy balances for each time step. For each step the heat energy input and the internal heat energy budget are calculated and applied to give a particular temperature profile. If the water is thermally (density) stratified, lifting work is required to mix a lower layer with the layer above it. The amount of energy required to lift a layer depends on the sharpness of the temperature gradient which determines the density differentials and the distance between the center of mass of the upper well-mixed layer and the center of mass of the layer to be mixed. The energy required to do the lifting work is derived from wind shear on the water surface. To determine if the energy supplied by the wind is adequate to mix an additional layer or if it is dissipated by viscosity with no additional entrainment, an energy ratio is used. When the ratio is greater than critical (Ford and Stefan, 1980; Stefan and Ford, 1975), i.e., the energy provided by the wind per unit volume of mixed layer is greater than the work needed to lift the layer below the mixed layer, then additional deepening of the mixed layer will occur.

To determine the heat input to the lake, net long wave (mostly atmospheric radiation which is absorbed at the water surface), net short wave radiation (mostly solar radiation, which is absorbed exponentially with depth with a specified attenuation coefficient), back-radiation, heat losses at the water surface by evaporation (condensation), and heat transfer by convection are all considered.

To adjust the incoming radiation at the water surface for reflection, an albedo of .06 was used. The cooling which occurs by evaporation at the lake surface is calculated using a relationship similar to that used by Brady, et al. 1969. The energy input by the wind is calculated from the shear stress at the water surface by a relationship proposed by Wu, 1969.

To model the Fairmont Lakes, the MLTM was modified so that weather data input and computations were carried out at a time step of 12 hours rather than 24 hours. The Fairmont Lakes are very shallow and respond more rapidly to meterological conditions than deeper lakes. The night cooling in these weakly stratified lakes induces mixing by natural convection, which plays a significant role. In the model the heat input is applied first, and then the wind energy input. The 12-hour time periods used were from 6 a.m. to 6 p.m. to 6 a.m. All the solar radiation was considered to occur during the 6 a.m. to 6 p.m. time period.

Solar radiation and wind velocity are the two most important weather variables in the MLTM model. Both may vary strongly from day to day and from year to year, and the response of the lake in terms of water surface temperature and vertical mixing is therefore very dynamic.

Simulations with the modified MLTM were made for five of the Fairmont Lakes under three observed summer weather sequences (1974, 1976, and 1977). The summer of 1974 was wet and cool and the summers of 1976 and 1977 were dry and hot. The simulation required weather data which were obtained from the Fairmont Airport, the Fairmont Municipal Water Filtration Plant, and the University of Minnesota Agricultural Experiment Stations at Lamberton and Waseca.

The simulations yielded vertical temperature profiles at 12-hour intervals. The results were represented graphically as mixed layer depths, as daily surface and bottom temperature, and as seasonal isotherm contours. Samples are shown in Figures 4, 5, and 1, respectively.

Prior to its application, the model was calibrated to minimize the standard error between measured and predicted water temperatures. The model calibration provided the constant reduction coefficient by which wind velocity data from the Fairmont Municipal Airport had to be adjusted to optimize agreement between measurements and predictions. That unusual procedure was adopted because the only available wind data were from an anemometer operated on top of an airport hangar, hence requiring adjustment.

After calibration, 19 of 22 identifiable midsummer lake overturns in 1974, 1976, and 1977 were accurately predicted.

An independent model verification was made after calibration by comparing predicted and measured

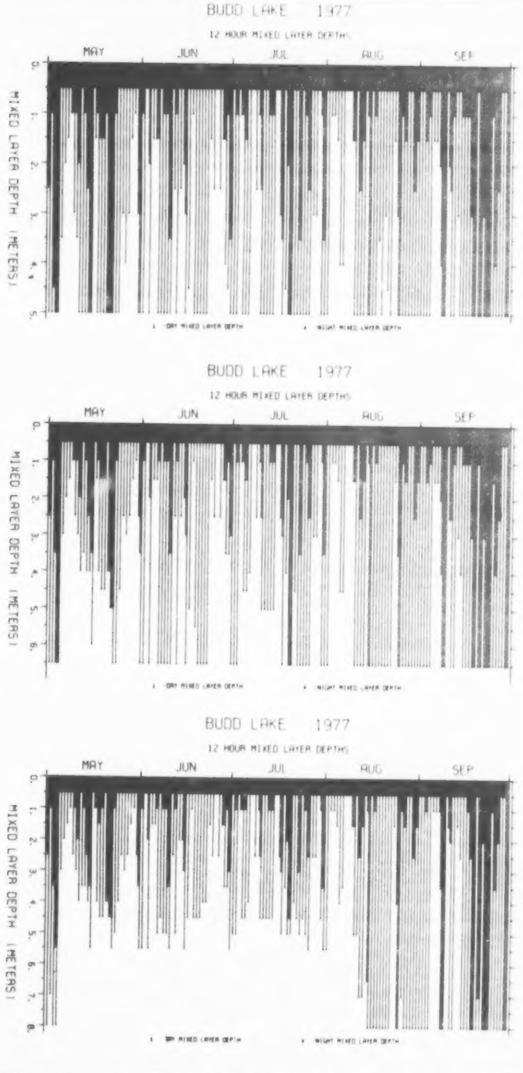

Figure 4. — Simulated mixed layer depths, Budd Lake, 1977. Maximum depth =5.0 m (top), 6.75 m (center and 8.0 m (bottom).

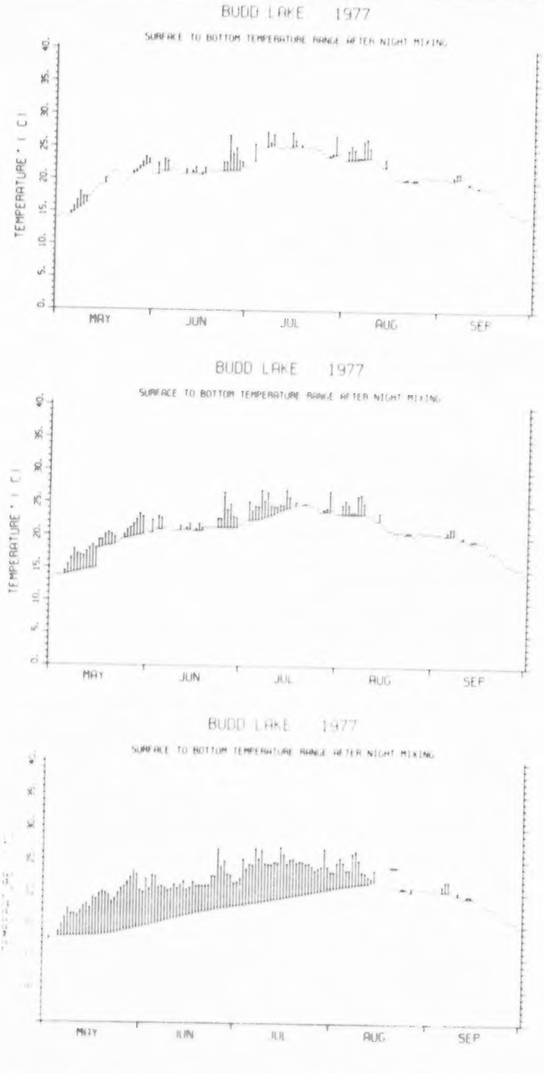

Figure 5. — Simulated surface and bottom temperatures, Budd Lake, 1977. Maximum depth =5.0 m (top), 6.75 m (center and 8.0 m (bottom).

water temperatures at a water withdrawal site (water treatment plant). The arithmetic mean of the temperature differences between prediction and measurements for a 3-year period was -0.41°C; the standard deviation was 0.95°C.

Dredging Effects on Stratification and Frequency of Mid-Summer Overturns

Lake temperature structures and mid-summer overturns were simulated for several post-dredging depths to determine the frequency of mid-summer overturns and the minimum depth for stable seasonal summer stratification (no mid-summer overturn). Model simulations were again made with the weather conditions encountered in 1974, 1976, and 1977. Different lake depth contours were used to simulate the effects of dredging.

Model results showed that under present conditions the lakes experienced several stratification periods separated by overturns during each of the three summers. Figure 4 provides detailed information on the simulated mixed layer depths in the morning and the evening of each day for Budd Lake under 1977 weather. By increasing the depth to 8 meters overturns can be prevented, as shown by the simulated plots of Figures 1 - and 4. A seasonal water temperature stratification lasting from the beginning of June through the end of August was required.

A summary of simulated frequencies of summer overturns and of stratification periods of 5 days or more is given in Figure 6 for the year 1974. A 5-day stratification period was chosen because field measurements of hypolimnetic dissolved oxygen concentrations showed that anaerobic conditions usually developed within 2 to 4 days, so that phosphorus release from sediments in the Fairmont Lakes can be expected to occur at the very latest after 5 days of stratification (Stefan and Hanson, 1979, 1980).

Undesirable intermittent stratification and multiple summer overturns which are a prerequisite for phosphorus recycling, were found most frequently in the depth range from 4 to 6 meters maximum lake depth. One of the lakes (Sisseton) mixed to the bottom in mid-June 1974, even when dredged to a maximum depth of 8 meters. A maximum depth of 9 meters was required to achieve a stable summer stratification in that lake.

Dredging to greater depths also decreases bottom water temperatures. Simulations of bottom water temperatures from Budd Lake during July 1974 were 25°C at 5 meters, 21°C at 7 meters, and 18°C at 8.5 meters maximum depth. The deeper lake conditions would therefore favor growth of game fish (i.e., walleye or northern pike) which are strained by low oxygen and temperatures above 22 to 23°C.

Recommendations for Required Dredging Depths

The information obtained by model simulations can be used to make recommendations for required dredging depths of the Fairmont Lakes based on mixing criteria:

1. Deepening the lakes to a maximum of 4 meters would be disadvantageous because it would increase the potential for sedimentary phosphorus release (increased number of periods of anaerobic conditions) and for transport of that phosphorus to the photic zone by overturns.
2. Dredging to maximum depths between 4 and 6.5 meters may be effective if the larger hypolimnetic water volume contains enough oxygen to prevent anaerobic conditions. From a mixing point of view, a maximum depth of 6.5 meters is not better than 4 meters.
3. A maximum dredged depth of 8 meters will effectively reduce phosphorus transport from the bottom to the photic zone (9 meters in Sisseton). Table 1 summarizes the anticipated improvements at different dredging depths.

The cost of dredging the Fairmont Lakes has been 72 cents per m³ in 1978 dollars. This figure ranks among the lowest quoted by U.S. EPA (1979). The anticipated total dredging costs for the Fairmont Lakes are given in Table 2. These costs are significant, but other restoration techniques such as inflow treatment, nutrient inactivation, and aeration would require continuous expenses and monitoring.

Compared to other lake improvement alternatives, dredging has a very lasting effect. Results of carbon-14

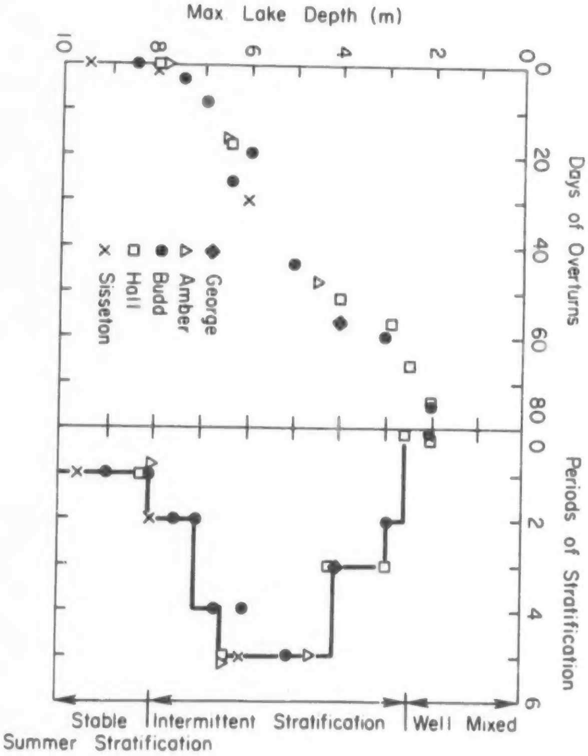

Figure 6. — Predicted number of days with overturns (complete vertical and mixing) and periods of stratification of 5 days or longer as function of maximum dredged depths from June 1 — August 31, 1974.

sediment dating suggest that over the last 9,000 years sediments accumulated in Hall Lake at the rate of 0.12 cm/yr. At this rate it would take about 420 years to refill 0.5 meters of dredged material.

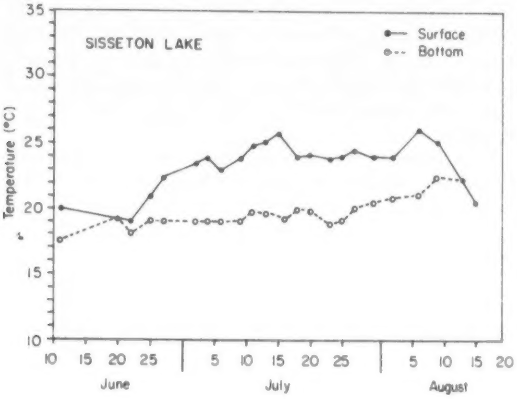

Figure 7. — 1979 surface and bottom temperatures in Sisseton Lake.

1979 Observations of Stratification Dynamics and Effects

In the summer of 1979, Budd and Sisseton Lakes were monitored two or three times a week. By a fortunate coincidence the lakes showed a seasonal stratification for the first time. They stratified toward the end of June and remained so until the middle of August, whereas in preceding summers several midsummer overturns had occurred. Unusually cold weather with substantial winds in the earlier part of June caused this occurrence. It was therefore possible to observe the phenomena which dredging is expected to produce annually.

Figure 7 illustrates the observed development and strength of the vertical temperature gradient. Figure 8 gives the observed dynamics of the mixed layer and the thermocline. Figure 9 illustrates the rise in ortho-phosphorus in the hypolimnion after its development and the absence of ortho-phosphate in the epilimnion. Associated chlorophyll *a* levels are shown in Figure 10. A significant drop in chlorophyll *a* occurred after the onset of stratification. There were no mid-summer blooms.

Seasonal stratification as experienced in 1979 maintained dissolved phosphorus and surface chlorophyll *a* at lower levels than in previous years.

The data shown in Figures 9 and 12 for Sisseton Lake and similar measurements in Budd Lake (Stefan and Hanson, 1979) agree with the hypothesis of phosphorus release and recycling from the sediments and the anticipated effects of dredging.

Table 1. — Budd Lake dredging diagnosis, 1974.

Maximum dredged depth (meters) (1)	Added depth (meters) (2)	Potential for anaerobic conditions and phosphorus release (3)	Number of mid-summer overturns (4)	Hypolimnetic water temperature in July (°C) (5)	Potential for algal blooms (6)	Water quality (7)
5.2	0	High	5	25-26	High	poor
6.75	1.5		4	21-24		
7.5	1.3		2	18-20		
8.0	2.8		0	17-19		
8.5	3.3	Low	0	16-18	Low	better

Table 2. — Predicted cost of dredging Fairmont Lakes to different depths.

Lake (1)	Surface area (km^2)	Present maximum depth (meters) (2)	Present average depth (meters) (3)	Proposed maximum dredged depth (meters) (4)	Proposed average depth after dredging (meters) (5)	Material removed (meters) (6)	Percent volume increase (7)	Cost at \$.72/meter^2 (8)
Amber	0.73	4.5	3.6	6.75	5.0	1.056×10^6	40	$ 760,000
				8.0	5.7	1.492×10^6	57	1,080,000
Hall	2.25	3.4	2.1	4.25	3.9	4.042×10^6	85	2,908,000
				6.75	5.6	7.863×10^6	166	5,657,000
				8.0	6.1	8.982×10^6	190	6,500,000
Budd	0.90	5.2	3.7	6.75	4.8	1.036×10^6	30	745,000
				8.0	6.0	2.069×10^6	62	1,497,000
Sisseton	0.54	6.0	3.5	8.0	5.8	0.844×10^6	43	611,000

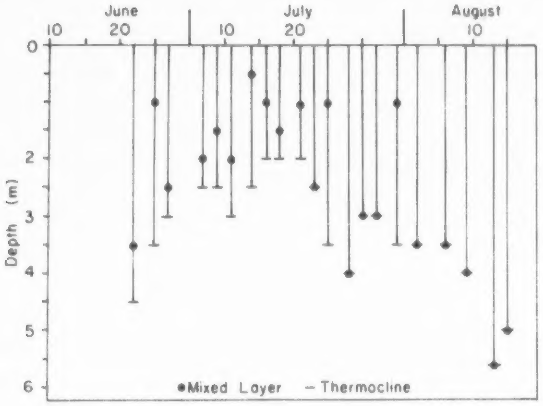

Figure 8. — 1979 mixed layer and thermocline depth in Sisseton Lake.

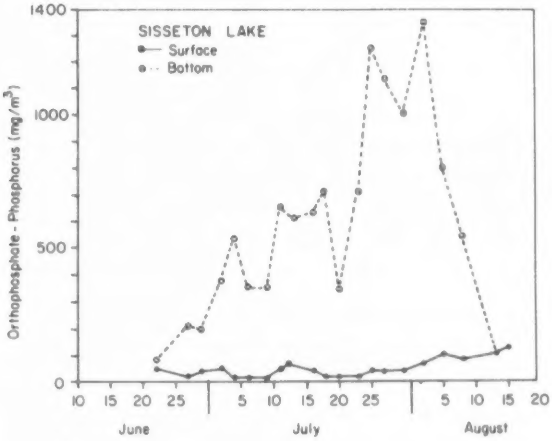

Figure 9. — 1979 surface and bottom orth-phosphorus concentrations in Sisseton Lake.

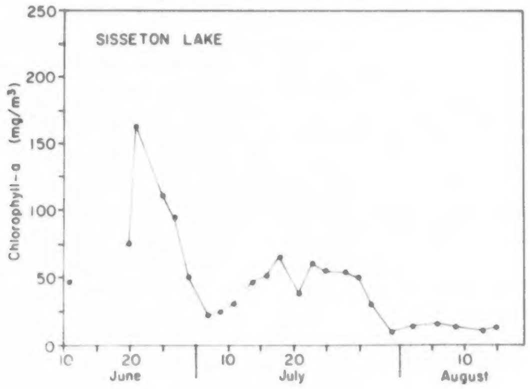

Figure 10. — 1979 surface chlorophyll *a* concentrations in Sisseton lake.

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DREDGING ACTIVITIES IN WISCONSIN'S LAKE RENEWAL PROGRAM

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ABSTRACT

Dredging has been the technique most often used in Wisconsin's lake renewal program. The program includes both natural and manmade lakes, with lake size and sediment removal up to 205 hectares and 1,720,250 cubic meters, respectively. A wide array of sediment type and disposal methodologies have been involved. Removal costs have ranged from 37 cents to $1.96 per cubic meter. During the dredging of Lilly Lake, water levels were lowered about 1.5 meters, resulting in a temporary reversal of groundwater flow. Dissolved oxygen levels were unchanged in the lake. However, there were increases in chlorophyll *a*, gross primary productivity, and some zooplankton species. Reductions occurred in water clarity and macrophyte biomass. Initial post-dredging monitoring indicates that groundwater inflow has increased greatly.

INTRODUCTION

There are nearly 15,000 lakes in Wisconsin, with a combined area of over 400,000 hectares. They form the foundation of the tourism/recreation economy, the third largest industry in the State. Citizen demand for better environmental protection of these lakes resulted in the creation of the Office of Inland Lake Renewal within the Department of Natural Resources in 1974. Subsequent rehabilitation projects have used various techniques such as aeration, dredging, drawdown, storm sewer diversion, aluminum treatment, aquatic macrophyte harvesting, improved animal manure handling, streambank erosion control, and several upland conservation methods. However, dredging has been the primary technique. This paper will describe: (1) The dredging program now underway; and (2) the Lilly Lake project.

DREDGING PROGRAM

Twelve projects are now in the program. Four are completed, two are currently underway, and six are finalizing plan proposals. Most of these lakes are manmade, originally created by dam construction. The lakes range in size from 4 to 205 hectares, with watersheds of 1.5 to 1,425 square kilometers. In some, the infilling rate exceeds 3 centimeters per year. Sediment removal varies from 26,760 to 1,720,250 cubic meters. Hydraulic dredging is the usual method of removal, but in four cases drawdown has been combined with lake bed excavation.

The sediment characteristics are diverse. In some cases the materials are dense, primarily sand, with a solids content of 70 to 80 percent. This is the usual situation when the lake is located on a major river system. At the other extreme, natural lake sediments are low density and organic. The solids content may be as low as 1 to 5 percent. Chemical composition is also variable, subject to previous lake and watershed usage.

Financial assistance to any project is dependent upon reasonable assurance of environmental improvement and permanency. Data collection is followed by predictive modeling and professional judgment to provide a basis for implementing a project (see Dunst, 1980 for further discussion). Removal costs have been higher for dryland excavation ($1.69 to 1.76/m^3) versus hydraulic dredging ($1.12 to 1.29/m^3) in projects of similar size (109,300 to 191,100 m^3). However, one large-scale hydraulic dredging project (683,900 m^3) has been undertaken to date with a per unit cost of only 37 cents.

Disposal site costs have been dictated by location, ownership, and usage. Projects have used settling basins (with or without allowance for return of carriage waters), low level diking, spreading on agricultural land, and spray irrigation. Wherever appropriate, erosion control practices have been applied in the watershed. These have involved primarily riprap, porous plastics, fencing, grassed waterways, diversion channels, contour strips, and conservation tillage.

Ongoing research activities include assessing: (1) The value of lake sediments on agricultural production; (2) the effect of dredging on a lake and associated groundwater system; and (3) alternative methods of lake deepening (organic sediments).

LILLY LAKE PROJECT

Lilly Lake is a natural, seepage lake located in southeastern Wisconsin possessing no surface inlets or outlets. The lake covers 37 hectares and in 1977 had a mean depth of 1.4 meters. Maximum water depth was 1.8 meters over more than 10.7 meters of organic sediments. The water content of the sediments ranged from 90 to 98 percent. The lake bottom and water column were filled with dense, rooted weed growth. Winter fishkills were common, and recreational opportunities severely restricted. Some fishing, boating, and swimming was possible in limited areas, but the recreational value was considered of poor quality. In-lake production and deposition were causing an infilling rate of 0.5 centimeter per year (e.g., radiometric dating using the Pb-210 method).

A project was undertaken to remove 683,900 cubic meters of sediment, increasing the maximum depth to 6.6 meters (Table 1). Dredging was initiated in July 1978 and continued until November. It commenced again in May 1979 and was completed by September. During 1978, a hydraulic dredge pumped approximately 382,000 cubic meters of sediment (or 798,200 m³ of sediment/lake water mixture) through a 30-centimeter diameter polyethelene pipe a distance of almost 3 kilometers to a settling basin. In 1979 the sediment was also applied to 15 hectares of agricultural land. A grant from the U.S. EPA has provided for monitoring the disposal sites and evaluating the effect of dredging on the lake. Evaluations include algae, macrophytes, invertebrates, fish, water quality, sediments, and groundwater. Investigations began in 1976 and will continue into 1982.

Table 1. — Depth — Water storage relationship before and after dredging; Lilly Lake.

	Before		After	
Depth (m)	Area (ha)	Accumulated volume storage (m^3)	Area (ha)	Accumulated volume storage (m^3)
0	37	532,300	37	1,216,200
0.6	32	320,500	35	995,800
1.2	29	132,200	33	787,100
1.8	14	0	31	590,000
2.4			30	403,700
3.0			28	228,600
3.6			9	115,700
4.2			6	70,900
4.8			5	40,300
5.4			4	15,400
6.0			1	2,300
6.6			.1	0

Measurements are being taken at least monthly during the summer. More frequent measurements were taken when the dredge was in operation. In-lake conditions before and during dredging are compared for the July/September period in Table 2. Dredging corresponded with increased chlorophyll *a*, gross primary productivity, ammonia -N, particulate phosphorus, conductivity, total alkalinity, turbidity, B.O.D. (5-day), *Bosmina*, and *Chydorus*; and decreased water clarity, soluble organic phosphorus, *Ceriodaphnia*, and macrophyte biomass. Also, as a result of sediment/water removal, the lake level was lowered about 1.5 meters. The groundwater system responded with increased flow into the lake around the entire perimeter (including reversal of previous outflow regions). This effect was verified by a network of monitoring wells and in-lake seepage meters.

Table 2. — In-lake conditions pre- and during dredging (July/September average).

Parameter	1976	1977	1978	1979
Chlorophyll *a* (μg/l)	2.5	3.3	18.5	9.5
Gross Primary Productivity (mgC/m³/day)	185	140	1005	-
Secchi disk (m)	est. 5	-	-	1.3
Macrophyte biomass (g/m²; dry weight)	685	335	-*	-*
Bosmina longirostris (#/1)	-	56	274	-
Chydorus sphaericus (#/1)	-	2	12	-
Ceriodaphnia sp. (#/1)	-	20	11	-
Dissolved oxygen (mg/l)	9.7	7.8	8.8	7.6
B.O.D. (5 day; mg/l)	-	1.6	3.6	-
Conductivity (μ mhos/cm at 25°C	-	247	317	433
pH (standard units)	-	8.3	8.0	8.1
Total alkalinity (mg/l as $CaCO_3$)	-	107	142	196
Turbidity (Formazin units)	-	1.2	3.0	4.1
Nitrite/nitrate -N (mg/l)	-	.02	.04	.05
Ammonia -N (mg/l)	-	.03	1.12	1.44
Organic -N (mg/l)	-	1.5	1.8	1.4
Soluble reactive phosphorus (μg/l)	-	4	4	4
Soluble organic phosphorus (μg/l)	-	16	6	5
Particulate phosphorus (μg/l)	-	17	30	19

* Biomass was nearly eliminated during dredging; based on visual examination.

The actual in-lake chlorophyll *a* levels in 1977 were closely predicted using Vollenweider (1976) and Sakamoto (1966) (Tables 2 and 3). Precipitation and groundwater inflow were much greater in 1978 (pre-July) versus 1977, and therefore these models were used to estimate the impact of changed climatic conditions alone on lake limnology. Mean phosphorus

Table 3. — Predicted chlorophyll *a* levels using Vollenweider (1976) and Sakamoto (1966).

*Condition:	1	2	3	4	5
Water loading (cfs.)					
Precipitation	0.31	0.47	0.34	0.34	0.34
Groundwater	0.07	0.31	0.09	0.09	0.31
TOTAL	0.38	0.78	0.43	0.43	0.65
Phosphorus loading (kg.)					
**Overland runoff	7.3	7.3	7.3	7.3	7.3
Precipitation	8.0	10.1	8.4	8.4	8.4
Groundwater	0.4	1.7	0.5	0.5	1.7
TOTAL	15.7	19.1	16.2	16.2	17.4
Predicted chlorophyll *a* (μg/l)	5.8	3.5	5.3	3.7	2.7
Hydraulic residence time (years)	1.5	0.7	1.4	3.1	2.0

*1. Actual measurements, 1977. Lake volume = 532,300 m³. Precipitation = 78 cm.
2. Actual measurements, 1978 (pre-July). Lake volume = 532,300 m³. Precipitation = 119 cm.
3. Theoretical normal precipitation year before dredging. Lake volume = 532,300 m³. Precipitation = 85 cm.
4. Theoretical normal precipitation year after dredging and assuming no change in groundwater inflow. Lake volume = 1,216,200 m³. Precipitation = 85 cm.
5. Theoretical normal precipitation year after dredging with expected increase in groundwater inflow. Lake volume = 1,216,200 m³. Precipitation = 85 cm.

**Based on published phosphorus loss coefficients for urban and forested lands.

levels in the areas of groundwater inflow were 6μg/l. Phosphorus values for direct precipitation (15 μg/l) and dry fallout (108 μg/ha/yr) were obtained from recent studies at a nearby location (Andren and Stolzenburg, 1978). Despite the increased water and phosphorus loading in 1978, chlorophyll *a* concentrations were predicted to decrease slightly. This provides further credence to the conclusion that dredging was responsible for the 1978 lake water quality.

In addition, sediment cores were collected in June 1978, and interstitial waters were analyzed for total dissolved phosphorus, ammonia -N, and nitrite/nitrate -N (Table 4). The sediments were obviously a major, potential source of ammonia -N and dissolved phosphorus. Earlier work with nutrient regeneration chambers on Lilly Lake in 1977 had also demonstrated an ammonia -N release rate of 21 to 32 mg/m^2 of lake bottom/day. Dredging would cause an increased impact of in-sediment conditions on lake water quality through physical disturbance of the sediments and greater transport in groundwater seepage, especially in the reversal region. These mechanisms apparently caused the higher in-lake ammonia -N levels. Some dissolved phosphorus was also being transmitted into the water column/biota; however, the impact was less pronounced. This probably resulted from oxidizing conditions present in the water column.

Table 4. — Sediment characteristics and interstitial water chemistry of a core from Lilly Lake.

Sediment depth interval (cm)	Total dissolved phosphorus (μg/l)	Ammonia-N (mg/l)	Nitrite/nitrate-N (mg/l)
0-8	105	2.1	<0.1
8-15	128	4.7	<0.1
15-23	118	6.7	<0.1
23-30	198	7.0	<0.1
61-69	202	12.0	<0.1
91-99	202	18.0	<0.1

* These sediments contained a solids content of 3% by weight.

Investigations are continuing into the post-dredging phase of this project. The lake water quality should revert to pre-dredging conditions. Early 1980 information indicates a permanent increase in groundwater inflow as a result of dredging. The lake's water storage capacity was increased by 128 percent while the hydraulic residence time appears to have risen by only 43 percent. This is a result of removal of low permeability lake sediments and an increase in the area of the lake bed (Beauheim, 1980). Although this impact will not be highly significant at Lilly Lake, the increased influx of low phosphorus groundwaters should have a beneficial effect on water quality (Table 3).

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NUTTING LAKE RESTORATION PROJECT: A CASE STUDY

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ABSTRACT

This paper is a case study of a 279,079 cubic meter lake dredging and watershed management program for Nutting Lake in Billerica, Mass. The restoration program, a 3 1/2 year effort, focused upon (1) lake dredging to deepen the lake to prevent reemergence of nuisance aquatic plants; and, (2) a watershed management program to reduce nutrient contribution to the lake from overland runoff. Funding was through EPA's 314 Clean Lakes Program, the Massachusetts Water Resources Commission's Research and Demonstration Program, and through cash and in-kind contributions from the town of Billerica. A consulting engineering firm, Purcell Associates of Hartford, Conn. was retained to refine the basic dredging concept, expand upon existing water quality data, design the containment facilities, conduct on-going water quality analysis, and evaluate the results. Two non-continuous operating containment basins with total capacity of approximately 91,752 cubic meters of dredged material were designed. The solids settled out while the remaining supernatant liquid decanted into small flocculation basins and then discharged into an outflow stream. This paper assesses the efficaciousness of various methods of treatment of the supernatant and compares the projected operating costs and dredged material production rates with those actually encountered during the first 2 years of operation.

INTRODUCTION

Nutting Lake is a 32-hectare lake located in a small, densely developed watershed in Billerica, Mass., situated in the larger Concord River watershed. Nutting Lake was a fashionable resort area for many Bostonians around the turn of the century. Today, one finds these summer cottages converted to year round housing to accommodate the suburban growth around Boston. This dense development, combined with year round use of seasonal houses, small, inadequate septic systems, and until recently, unpaved roads, has reduced Nutting Lake to little more than a large sediment basin. Because of the highly eutrophied state of the lake, boating, fishing, and swimming were virtually nonexistent when this project got underway in 1977.

Physically, chemically, and biologically, Nutting Lake has many characteristics of an urban and eutrophying lake: Dense watershed development; unused or underused recreational potential; high nutrient concentrations from septic and overland runoff; and submergent and emergent macrophyte growth. The lake itself is bisected by the Middlesex Turnpike, creating two basins; one is 11 hectares, the other 20 hectares. Both basins, connected by a 1.5 meter culvert, have the same mean depth of 1.3 meters, and maximum depths not exceeding 2.1 meters, which makes for a total volume of 41 hectare-meters. With its high surface-to-volume ratio and its small size and shallowness, the lake is highly productive.

There is one major inlet and one outlet to Nutting Lake. The inlet in the northeast corner of the east basin has very low flows, is dry during the summer months, and is supplemented by seepage from several swampy areas that surround the lake. The outlet, at the west end of the west basin flows into Mill Brook, and eventually into the Concord River. U.S. Route 3, a four lane divided highway, passes almost directly over the outflow as it leaves the lake.

The lake occupies a shallow depression in bedrock that is overlain with a thin layer of glacial till and debris. Depth to bedrock is generally ½ to 3 meters, with frequent rock out-croppings. The soils are well drained, and are not conducive to supporting on-lot septic systems.

Until 1975 these septic systems, often of inadequate size, were the sole means of sewage disposal. In 1975, interceptor sewers were provided. By 1980, the town required connection to the system. As recently as 1979, 45 percent of the residential dwellings and 32 percent of the dwellings that front directly on the lake were unsewered. While there are several plausible arguments that suggest greater compliance than these numbers show, it remains that the septic system problems have been a major contributor to declining water quality.

Major discharge points into the lake along its north shore drain an area of approximately 8 hectares, or 3 percent of watershed. In addition to direct stormwater discharge at these four points, uncollected runoff from streets and lots accounts for an unquantified but significant source of additional input.

WATER QUALITY

Water quality information was gathered by the Massachusetts Division of Water Pollution Control in 1975; baseline conditions were further analyzed before the dredging program began. Similar analytical methods and sampling locations were used in the most recent baseline program so that consistent information is developed by which water quality changes can be monitored.

The water chemistry of Nutting Lake observed in 1975 and 1978 indicated the lake's eutrophic state. Total phosphate concentrations ranged from 0.02 to 0.39 milligrams per liter (mg/l) as P; higher concentrations were measured during the winter and early spring periods. Ammonia nitrogen concentrations from the various sampling locations ranged from a low of 0.00 mg/l in the 1975 study to 1.30 mg/l in the 1978 study. Color values were high, as expected. Secchi disk visibility, during high algae growth periods, was limited to approximately less than 1/3 meter.

Bacteriological quality information pointed to the contributions of both stormwater and direct discharge. Elevated coliform levels during storms were measured in the spring and fall of 1977. Fecal coliform levels in the stormwater, when measured with respect to time, dropped to 12 percent of their initial value 1 hour after the onset of the storm, leading to the conclusion that some sewage was present in the stormwater. Furthermore, field observations and aerial photographs in 1977 indicated that raw sewage was reaching the lake, particularly near its inlet in the east basin.

The greatest problem in Nutting Lake was, and continues to be the presence of algal blooms and floating and rooted macrophytes. Submergent and floating macrophytes were most abundant throughout the lake.

The submergent group, represented by bladderwort and pondweed, were concentrated in the west end of the west basin, near the outlet, though they were present in fewer numbers throughout the lake. Floating macrophytes, also concentrated near the outlet of the west basin, were abundant elsewhere in the lake, particularly along Middlesex Turnpike. Represented primarily by watershield and yellow lilies, they accumulated along the shores of the south coves suggesting they were strongly influenced by the prevailing westerly wind, allowing proliferation in the more protected areas.

Emergent macrophytes were not significant, either in species represented or areal extent, as the emergent or floating variety. This is mainly because of the extensive shoreline development.

During the initial phase of the restoration program, prior to dredging, macrophytes were identified first from a boat, then using an aerial high speed black and white and color infra-red imagery. They were then mapped for use in designing a dredging program.

Plankton samples were also taken during the initial phases of the restoration program. Employing standard methods of sampling and identification, enumeration was done qualitatively as well as quantitatively. Both the east and west basins exhibited high algal counts as would be expected with high nutrient loadings. Two blue-greens, *Aphanizomenon* sp. and *Anabaena* sp. dominated in the east basin during the baseline survey period, while *Aphanizomenon* sp. was the only dominant blue-green in the west basin.

Subsequent sampling of the basins in the following years coupled with aerial reconnaissance flights has pointed to an interesting phenomenon: algal blooms were occuring first in one basin, then in the other basin the following year. This pattern, which has been observed for 4 years, will be discussed later in this report.

PROGRAM DEVELOPMENT

In 1977 the Billerica Conservation Commission, with the assistance of the Northern Middlesex Regional Planning Commission, sought a 314 Clean Lake Grant from the U.S. Environmental Protection Agency. The goal was to remove nutrient-laden sediment and macrophyte growth. They also hoped to deepen the lake by dredging. Coupled with this in-lake program was a watershed management program. It was primarily aimed at controlling nutrient input from runoff by street-sweeping and limiting further watershed development by land acquisition. At the initial stages, approximately 164,389 cubic meters of material was to be removed from the lake by using a Mudcat hydraulic dredge, and deposited in a basin on town-owned land on the opposite side of U.S. Route 3.

With the basic program concept of dredging and watershed management approved by EPA , the town sought financial assistance from the Massachusetts Division of Water Pollution Control and the Massachusetts Water Resources Commission. Agreeing to participate in the project and fund the demonstration of dredge material disposal, the Water Resources Commission contracted with Purcell Associates of Hartford, Conn. to refine the dredging program, conduct baseline water quality analyses, design a dredged material containment area, and evaluate the entire restoration project on completion.

At the same time, the town raised funds at a town meeting to match the EPA and State contributions. More importantly, the town agreed to supply dredge operators for the duration of the project as an in-kind service.

In the summer of 1977, baseline water quality analysis, vegetation and biological analysis, dredging area refinement and preliminary design of the dredge material disposal areas were all undertaken by the consultants. A Phase 1 report was then submitted to the State, the EPA, and the town, with recommendations on dredging areas and dredged spoil containment and disposal methods.

DREDGING AREAS

The Phase 1 report concluded with a recommendation of expanding the amount of bottom sediment to be removed from 164,389 cubic meters to 279,079 cubic meters. It also recommended expanding the initial 2 year program to 3½ years to accommodate the increased volume of material to be dredged. There were several reasons for the decision to expand the dredging program:

1. Analysis indicated that the very fine bottom sediments suspended in the water column by the wind drag further contributed to color and turbidity problems;
2. Removal of the existing macrophytes, including their root systems;
3. Increased removal of the benthic nutrient store;

4. Reduction of the substantial source of oxygen-demanding organics; and

5. Enhancement of lake aesthetics.

Based on this expanded program, a 5-day, 10-hour/day dredging program was designed, so that dredging could be accomplished in the suggested period. This would result in deepening Nutting Lake from an average 1.3 to 2 meters. The dredging, which was to be carried out by town workers under auspices of the Conservation Commission, was to focus initially on the dense macrophyte beds in the outflow region of the west basin. Dredging was to be done to prescribed depths at this area, so as to investigate macrophyte regrowth as a function of lake depth. The preliminary results of this experiment are discussed later in this report.

DREDGED MATERIAL DISPOSAL

One of the most critical operational components of any lake dredging project is the proper disposal — both from an environmental and an engineering perspective — of the dredged spoil. There are certain elements that must be considered:

1. Selection of a suitable spoil area within pumping capacity of the equipment;

2. The containment area must be designed and operated so that the dredged material can be received and dewatered at as high a rate as possible, and so that the supernatant can be discharged at an acceptable level of quality; and

3. The containment area must be reusable once the project has been completed.

At Nutting Lake there was a suitable site available within pumping distance and outside the watershed.

Designing a dredged material containment as well as establishing operation procedures for the area is a function of the amount of material to be placed in the containment area during a dredging season, the dredge production rates, the settling and bulking characteristics of the dredged material, and the environmental restrictions placed on the quality of the discharged effluent. By using manufacturer-supplied information and field observations of past projects, dredge production rates from the Mudcat were projected and related to the 10-hour work day chosen by the town. Because the town chose to work only 10-hour days, a noncontinuous mode of operation at the containment area was employed, to allow for a quiescent settling period prior to discharging the supernatant.

To size the containment basins, core samples of the lake bottom sediment were obtained from both basins, and their settling characteristcs were simulated in the laboratory. This was done by mixing the sediment with lake water (achieving a solids content approximately the magnitude of that pumped by the Mudcat) and pumping the slurry into a 1.8 meter column tube and allowing it to settle. The rate of settlement for the solid/liquid interface was charted, and a 90 percent settlement of the suspended material was achieved after 6 hours and 40 minutes. Thereafter, there was no significant increase in settlement. Based on the lab results, a containment area detention time of 7 hours was established.

Further settlement column tests were run at different heights to estimate sediment consolidation caused by self-weight stresses. Sediment volume and water content were first measured in the core tubes, and then subsequent to sedimentation in the column testing tubes. From these two measurements the bulking factor (here defined as the ratio of a given volume of the same amount of solids on the lake bottom) was determined for the sediment from each basin. For both basins the bulking factor, as calculated from these tests, ranged from 1.2 to 1.6, depending upon the effective stresses. Obviously, as more solids are deposited into the basin the self-weight stresses increase, further consolidating the material, and resulting in a lower bulking factor. Based upon these tests, a final solids height (approximately 30 percent of the initial slurry height) was used as a design parameter.

Based upon the town's dredging schedule and a 7-hour quiescent settling period, preliminary design of the containment area was initiated using the noncontinuous mode of operation. While the Billerica Conservation Commission has shortened the detention times at no loss in water quality, the original design called for a 7-hour settling period and, prior to initiation of dredging the following day, draw-off of the supernatant using an adjustable outflow device.

The disposal area site made available by the town was a 7-hectare wooded parcel west of Route 3, approximately 152 meters downstream of the outflow of Nutting Lake, adjacent to Mill Brook. The topography at the site was such that, while it was somewhat limiting, the slopes could be used to advantage in designing the containment area. Several designs were developed by the consultants. The one selected consisted of a two basin design, one of 65,000 cubic yard capacity and one of 63,000 cubic yard capacity. These had capacity sufficient for a season's dredging. At the end of each dredge season the material is removed and initial capacity is restored.

Though no specific EPA effluent standards existed at the time, treatment of the supernatant prior to discharge to Mill Brook was mandatory. The column settling tests indicated that settling periods of up to 12 hours produced no appreciable improvements in supernatant turbidity beyond the planned 7-hour detention time. These turbidity levels were 100 nephelometric turbidity units (NTU) at both 6- and 12-hour intervals. Settlement times in excess of 3 days only reduced turbidity to 50 NTU. While the 50 NTU level met the agreed-upon guideline, the 3-day settling period was unacceptable. As a result, several treatment alternatives were investigated for cost as well as their effectiveness in meeting the standard. One alternative consisted of pumping the supernatant into a fabric filter basin and draining it through the sides of the basin into a perimeter swale for discharge into Mill Brook. This method proved ineffective in improving supernatant quality, although several weaves of fabric were used in these tests.

A second treatment alternative consisted of discharging the supernatant into the wetlands bordering the lake in the northwest portion of the west basin. A detailed site investigation revealed a typical Massachusetts wetland with low pH values ranging from 4.1 to 5.3.

Although sedimentation and filtration in the wetland would facilitate the removal of supernatant turbidity, concern existed that the low pH might release various metals that might otherwise remain in a bound state, or that the pH of the wetland might be altered. Therefore, this treatment alternative was rejected.

A third alternative involved decanting the supernatant from the containment area to a flocculation basin. A low molecular weight cationic polymer would then be introduced by metering pump into the discharge pipelines at the outflow from the containment area. The supernatant would then be held in a quiescent state in the flocculation basin prior to its discharge into Mill Brook.

Jar testing was done, using polymers at various concentrations to determine the optimal concentration to achieve the desired clarity. A range of 8 to 10 parts per million was found to be sufficient to achieve removal levels that permitted direct discharge into Mill Brook. It was this method of treatment that was chosen, a system that is functioning adequately after 2½ years of dredging. It is interesting to note that turbidity measurements at the flocculation basin outflow average 2 NTU, and compare favorably with a turbidity measurement of 9 NTU in Mill Brook and with the 1 to 5 NTU's for Nutting Lake itself.

PROGRAM COSTS

Before discussing water quality results and the efficacy of this program, it is useful to review the costs, on a per unit basis. In a project such as this, two of the three biggest cost items (purchase or lease of a dredge and construction of a disposal area) are fixed costs. Their unit value is inversely proportional to the quantity of dredged material removed. Labor costs, the third high-cost item, are variable, and generally proportional to the quantity of material removed. Based on removal of 279,079 cubic meters during a 3½ year period, estimated costs for the dredge, containment areas, operation and maintenance and labor come to approximately $522,000, or approximately $1.87 cu/meter. By most standards $1.87 cu/meter ($1.45 cu/yd) for removal and disposal is an excellent price and compares most favorably with costs encountered on other dredging projects. Parenthetically, it is important to mention two things: Estimates of removal rates are based upon information supplied by the dredge manufacturer, as well as field observation, and the town may be a little behind schedule which would increase labor and operation costs. Secondly, labor costs are not artificially depressed; they are based upon hourly rates paid Department of Public Works employees by the town.

Perhaps the element that appears most promising is the fact that contractors are willing to pay for the dredged material. As part of the Phase 1 report, a detailed chemical analysis of the sediment was done to determine if it has reuse value. Reuse of the material, primarily as a soil conditioner, had been planned from the beginning of the project, and earlier this year the Conservation Commission let a contract for the purchase of 152,920 cubic meters (200,000 cubic yards) of material at a price of $1.40 cu/meter ($1.15 cu/yd). This would provide the project with a revenue stream of $215,000 which, projected over the life of the project, would lower the unit cost well below $1 per unit measurement.

CONCLUSION

Greatly improved water quality has not resulted to date. Before the wrong conclusion is drawn, however, it should be said that sufficient dredging has not yet occurred to produce dramatic water quality improvement. We are optimistic that with sufficient sediment removal (in the west basin dredging has taken place down to a gravel bottom) the annual cycle of self-fertilization will have been broken, and one of the primary sources of nutrients will have been removed.

Of concern is the possible shift from a lake dominated by macrophytes to one dominated by algae. As alluded to earlier, an interesting algae situation exists in Nutting Lake, which has been monitored for the last 4 years. In 1977 and 1978 physical evidence and observation indicated the dominance of blue-greens (*Aphanizomenon* sp.) in the east basin with an average areal biomass of 18,855 ASU/ml versus 6,770 ASU/ml in the west basin. However, 1979 data contradict the earlier observations and measurements by the Division of Water Pollution Control that indicated that areal biomass in the west basin is approximately four times greater (3,845 ASU/ml versus 13,795 ASU/ml) in the east basin. There are several possible explanations. The reversal may be because autumn blooms occur at separate times in each basin, and perhaps are dynamically related, or that sampling was conducted at different stages of bloom development. Since significant changes appear to be occurring during September and October increased sampling frequency will be employed this year in an attempt to deduce the cause.

From an operations perspective the project has been very successful. Once the initial bugs were worked out of the system, dredging proceeded smoothly. There are several small design changes that should be incorporated into similar projects, particularly in the areas of weir design and embankment stabilization. The Conservation Commission's experience with effluent treatment and shortened detention times is evidence that smaller basins could be employed, thereby reducing construction costs and the land area required for disposal.

Operator efficiency, a concern whenever labor is employed, has improved dramatically with experience. Down time has been reduced and dredge production rates equal or exceed the rates supplied by the manufacturer. With additional water quality information still forthcoming, (information that will be critical to the final evaluation of the success of dredging as a restoration technique) it can be said that the viability of dredging as a restoration has been at least partially demonstrated and the Nutting Lake project can serve as a model for municipalities and/or lake restoration practitioners who are contemplating similar projects.

MERCURY SPECIATION AND DISTRIBUTION IN A POLLUTED RIVER-LAKE SYSTEM AS RELATED TO THE PROBLEM OF LAKE RESTORATION

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ABSTRACT

Available techniques for preventing mercury pollution and restoring mercury-polluted inland waters are reviewed, and the feasibility of applying them to the Wabigoon River system of Northwestern Ontario (Canada) is considered. The Wabigoon River and an associated chain of lakes are polluted with mercury from a chlor-alkali plant/paper mill complex. Methyl mercury (CH_3Hg^+) levels in surficial bottom sediments depend on environmental factors but are unrelated to total Hg concentrations. CH_3Hg^+ content is related to pH, nutrient supply, and microbial methionine biosynthesis in wood-chip deposits near the source of pollution but appears to be a function of sorption, complexation, and precipitation phenomena involving Fe and Mn oxides, cation exchange sites, humic matter, and sulfide in clay-silt muds further downstream. Sedimentary CH_3Hg^+ levels are relatively high throughout the system despite an exponential decrease in total Hg downstream from the industrial complex. In the wood-chip sediments CH_3Hg^+ production is most intense at the sediment-water interface even though total Hg is most abundant below the interface. The river water, and therefore surface waters of lakes through which the river flows, are continually being contaminated by CH_3Hg^+ released from these deposits. CH_3Hg^+ is also released into hypolimnion water from Hg-contaminated lake mud. Pelagic fish (e.g., walleye) in the lakes are probably contaminated principally by CH_3Hg^+ introduced by the river. Only bottom-feeding animals (suckers and crayfish) seem to be strongly affected by CH_3Hg^+ formed locally in the lake sediments. Consequently, Hg in the riverbed above the lakes must be removed or immobilzed (e.g., by dredging or by accumulation and treatment in settling ponds) to reduce Hg concentrations in the fish species of primary importance to fishermen. Attempts to ameliorate the lakes without taking the constant influx of contaminated river water into account would probably be unsuccessful.

INTRODUCTION

Pollution of natural waters by mercury (Hg) can be prevented by removing Hg from effluents and restricting its use. However, there have been virtually no attempts to restore bodies of water already contaminated with Hg, although potentially useful procedures have been tested in laboratory and field experiments.

In any attempt to reclaim a contaminated system, the chemical speciation as well as the total quantity and distribution of the Hg must be considered. Although Hg in bodies of water is mostly in the form of Hg^{2+} ions bound to sediments, the harmful effects of the Hg are due principally to formation of monomethyl mercury (CH_3Hg) from the Hg^{2+} by free-living microorganisms (Fagerstrom and Jernelov, 1972). This water-soluble, yet fat-soluble, compound is released into the water and is readily accumulated by fish, whose meat may thereby be rendered poisonous to human consumers. Rates of synthesis and release of CH_3Hg^+ are determined by environmental variables such as pH, E_h, nutrient supply, and the abundance of sulfide and other Hg-binding agents.

This report reviews the available pollution control techniques and shows how some of them might be applied in a specific problem region: The Wabigoon-English-Winnipeg River system of Canada.

METHODS FOR EFFLUENT PURIFICATION AND RESTORATION OF LAKES AND RIVERS

Effluent Purification

Heavy metals can be removed from effluents at their source (Bell, 1976) or at regional treatment centers, as in Switzerland (Anonymous, 1976), and then recycled or stored. Methods suitable for Hg include the following:

1. *Clarification*, whereby particulate Hg is allowed to settle out of turbid wastewaters.

2. *Chemical treatment*, whereby dissolved Hg is precipitated or is scavenged by adsorbants: Organic and inorganic sulfides (e.g. S^{2-}, FeS, pyrite (FeS), alkyl thiols, and wool fibres) are particularly effective (Feick, et al. 1972; Suggs, et al. 1972; Tratnyek, 1972; Jernelov and Lann, 1973; Reimers and Krenkel, 1974; Chow and Buksak, 1975; Brown, et al. 1979). Other binding agents include elemental sulfur (Suggs, et al, 1972), peat (Feick, et al. 1972), Fe and Mn oxides (Lockwood and Chen, 1973; Kinniburgh and Jackson, 1978), and nitrogenous polysaccharides and complex amines (Moore, 1972; Snyder and Vigo, 1974). Recovery of Hg^{2+} as Hg° by reaction with Al° (Maag and Hecker, 1972) and co-precipitation with wastes

flocculated by aluminum sulfate (Jernelov and Lann, 1973) have also been proposed.

3. *Accumulation of Hg* by batch cultures of algae or other organisms followed by filtration or other procedures for harvesting the cultures (Filip, et al. 1979).

A method employing sewage for production of both algal blooms and biogenic sulfide in settling ponds has been proposed for removal of Hg and other heavy metals from effluents (Jackson, 1978).

Lake-river restoration

Restoration procedures must be selected to suit the individual environment. At best, these methods entail problems such as high cost, harmful side effects, technical difficulties, and limited effectiveness. Thus, the benefits and disadvantages must be carefully weighed. The following techniques have been considered:

1. *Physical removal of contaminated sediments by dredging or pumping, and other mechanical operations* such as ploughing bottom sediments (to dilute contaminated sediment with underlying "clean" sediment), river diversion, and manipulation of river flows to flush contaminated sediments into holding ponds (D'Itri, 1972; Feick, et al. 1972: Jernelov and Lann, 1973; Jernelov, et al. 1975; Parks, et al. 1980; Wilkins, 1980; Wilkins and Irwin, 1980). Difficulties include high cost, possible resuspension of sediment (leading to wider dissemination of Hg and acceleration of methylation rates), and the problem of providing for safe disposal of dredge spoil (to avoid contamination of other environments).

2. *Chemical treatments and biomanipulation*: (a) Maintenance of mildly alkaline to neutral pH levels in the water and sediments (e.g. by adding lime or reducing industrial SO_2 emissions to control "acid rain") to favor production of dimethyl mercury ((CH) Hg) rather than the more pernicious CH_3Hg^+ (Jernelov and Lann, 1973; (b) maintenance of reducing conditions and high rates of sulfide genesis in bottom sediments, and eutrophic conditions in the water column (e.g., by fertilization of lakes), to suppress formation and release of CH_3Hg^+, and also to dilute the CH_3Hg^+ with a large, rapidly growing biological community (D'Itri, 1972; Jernelov and Lann, 1973; Jernelov, et al. 1975); (c) addition of selenium compounds to the water to detoxify Hg and inhibit Hg accumulation by fish (Ganther, et al. 1972; Rudd, et al. 1980); (d) miscellaneous biomanipulations such as promoting growth of demethylating bacteria, suppressing formation of CH_3-cobalamin (a factor in microbial Hg methylation), removing Hg from fish protein, fostering bacterial conversion of Hg(II) to Hg°, and using batches of Hg-scavenging organisms (Wood, 1971; D'Itri, 1972; Spangler, et al. 1973). At best, none of these methods would be fully effective, and some could be harmful. Method (b) would involve undesirable side effects, while method (c) requires further research and extreme caution, as selenium itself can be toxic. The procedures listed under (d) are merely hypothetical possibilities.

3. *Covering contaminated sediments* with a layer of sand, gravel, silt, clay, silica, nontoxic mine tailings, sulfide minerals, elemental sulfur, or even sheets of manmade materials such as plastic to inhibit methylation and retard the release of Hg species into the water (Bongers and Khattak, 1972; D'Itri, 1972; Feick, et al. 1972; Suggs, et al. 1972; Widman and Epstein, 1972; Jernelov and Lann, 1972; Langley, 1973). Possible difficulties include disruption of protective sediment layers by gas bubbles, current action, or benthic animals. The use of plastic sheets would be costly and could have harmful ecological effects.

3. *Purification of Hg-polluted water bodies by natural processes*: Once the input of anthropogenic Hg is halted, the Hg in the sediments is eventually sealed off by layers of uncontaminated sediment or dispersed by mechanisms such as current action and evaporation of volatile species. In most cases, however, it would take an excessively long time — several decades at least, and possibly centuries — for a system to be fully restored by natural processes alone (Langley, 1973; Jernelov, et al. 1975).

MERCURY POLLUTION IN THE WABIGOON-ENGLISH-WINNIPEG RIVER SYSTEM

Introduction

The Wabigoon-English-Winnipeg River system flows northwestward over a distance of 420 kilometers through a chain of lakes extending from Wabigoon Lake (Ontario) to Lake Winnipeg (Manitoba). The system is situated in a sparsely populated region of boreal forest, low relief, and Precambrian rocks overlain by patches of Pleistocene deposits. This report principally concerns the Wabigoon River together with Wainwright Reservoir, Clay Lake, and Ball Lake, through which the river flows, in that order, after passing the town of Dryden, where the pollution orginated (Figure 1).

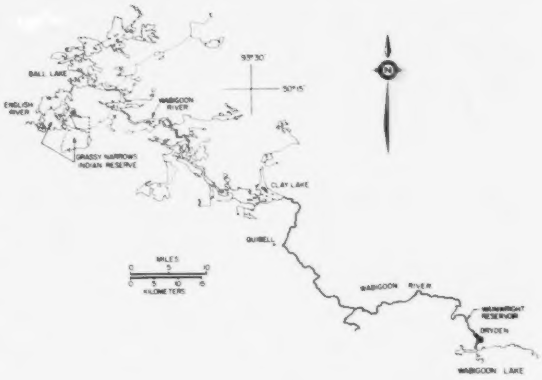

Figure 1. - Map of the Wabigoon-English system from Wabigoon Lake to Ball Lake.

Between 1962 and 1970 uncontrolled quantities of inorganic Hg amounting to about 10 metric tons were released into the Wabigoon River from the chlor-alkali plant of a pulp and paper company (known at different times as Dryden Paper, Reed, and Great Lakes Forest

Products, Ltd.) at Dryden (Armstrong and Hamilton, 1973). Large volumes of wood fragments and other paper mill wastes, as well as treated sewage, have also been discharged into the river (German, 1969). Under pressure from the Ontario government, Hg discharges were reduced in 1970 and supposedly halted in 1975 (Bishop and Neary, 1976) but have, to an appreciable extent, continued to the present day (Parks, et al. 1980). Anomalously high Hg levels have been found in the top 5 cm of sediments as far as 240 kilometers downstream from Dryden (Parks, 1976), and fish at least 270 kilometers downstream have Hg concentrations above the background levels found in nearby unpolluted lakes, and well above the 0.5 ppm legal limit for edible fish marketed in Canada (Fimreite and Reynolds, 1973; Bishop and Neary, 1976).

The chief human victims of the Hg pollution are the Ojibway Indians of the Grassy Narrows and White Dog bands. They have suffered economic hardship because commercial fishing in the region had to be banned, and some of them have high blood levels of Hg owing to consumption of locally caught fish. Neurological abnormalities which may well be due to mild methyl mercury poisoning have been detected in several tribesmen (Troyer, 1977; Wheatley, 1979).

The present study concerns the biogeochemistry and distribution of Hg species in the Wabigoon River system and implications for restoration of the system. This as yet unfinished work is reported in greater detail elsewhere (Jackson and Woychuk, 1980); it has been incorporated into a more extensive group project undertaken jointly by the governments of Canada and Ontario (Jackson, 1980).

Methods and materials

Water samples and grab samples or cores of bottom sediment collected in the spring and summer of 1978 were analyzed for CH_3Hg^+ by gas chromatography after extraction with benzene or toulene, respectively, and $NaBr/H_2SO_4+CuSO_4$. Total Hg was determined by flameless atomic absorption after digestion with H_2SO_4/HNO_3 at 160°C (in the case of sediments) or $KmnO_4/H_2SO_4$, $K_2S_2O_8$, and ultraviolet radiation (in the case of water). The sediments were analyzed for pH, E , free and "bound" (nonvolatile, 6 N HCl-soluble) sulfide (S^{2}), organic carbon (org. C), nitrogen (N), iron (Fe), manganese (Mn), $NH_2OH{\cdot}HCl/HNO_3$ and citrate/bicarbonate/dithionite (CBD)-extractable Fe and Mn, and amino acids (following acid hydrolysis). Following centrifugation and rinsing with nitrogen-purged water, the sediments were extracted with various nitrogen-purged solvents such as Ca acetate and $NH_2OH{\cdot}HCl/HNO_3$ to isolate different operationally defined Hg species. Data for sediments were based on oven-dry (105°C.) weight.

Results

Total Hg concentrations in the surficial bottom sediments of the Wabigoon River system decrease exponentially with distance downstream from Dryden, owing to progressive attenuation of Hg-contaminated detritus (Figure 2, A & B). Similarly, these sediments show a downstream decrease in organic C and N accompanied by an increase in the N/org. C ratio and an increase in pH (within the range 4.40 to 7.30), reflecting a gradational change from deposits of putrefying wood fragments near Dryden to clay-silt mud associated with humic matter and Fe-Mn oxides further down the system (Figure 2C). In contrast,

Figure 2. — Variation of bottom-sediment geochemistry with distance downstream from the industrial complex at Dryden A. Total and methly mercury data for the Wabigoon River between Dryden and Quibell. B & C. Mean values of total and methyl mercury, "bound" sulfide, and organic carbon content, and nitrogen/carbon ratio, for principal depositional basis from Dryden to Ball Lake.

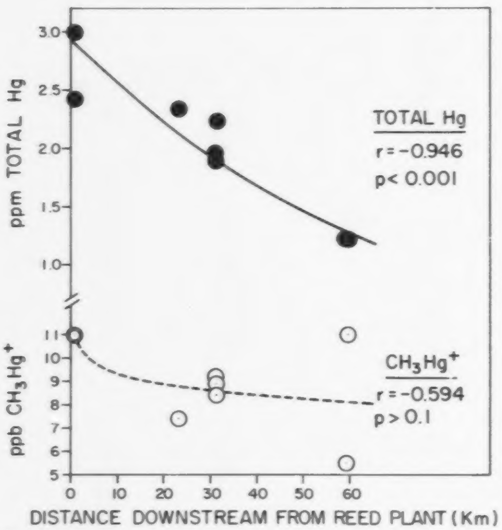

Figure 2A

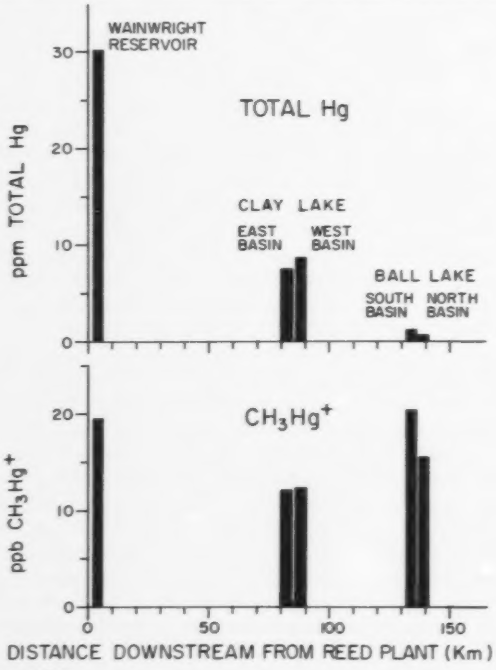

Figure 2B.

CH_3Hg levels show no significant long-range variation with distance (Figure 2, A & B). The slight depression of CH_3Hg^+ levels in Clay Lake can be attributed to anomalously high sulfide concentrations (Figure 2C) resulting from eutrophication caused by the influx of riverborne nutrients. CH_3Hg^+ production in the sediments is evidently limited by environmental factors rather than total Hg supply.

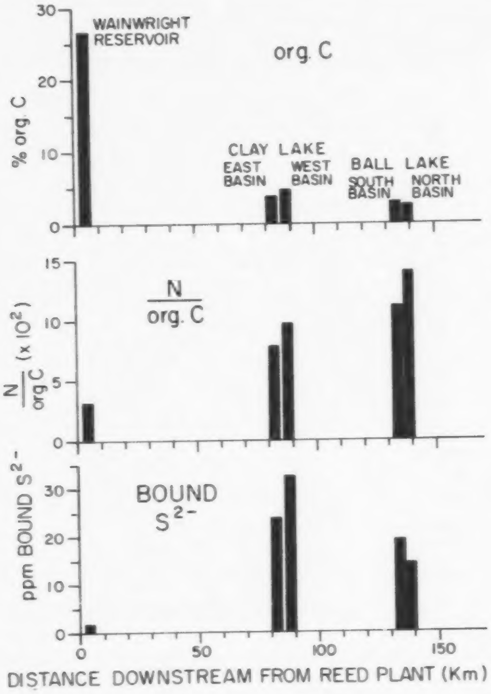

Figure 2C.

Cores from Wainwright Reservoir and the riverbed above it showed that CH_3Hg^+ concentrations were generally highest at the sediment-water interface (Figure 3), presumably reflecting particularly favorable conditions for microbial CH_3Hg^+ production, despite the tendency of Hg-enriched deposits laid down during 1962 to 1970 to be buried by post-1970 sediments of lower total Hg content. Production of CH_3Hg^+ at the sediment-water interface leads to continual release of CH_3Hg^+ into the overlying water, followed by fluvial transport into the surface waters of Clay Lake and probably Ball Lake. This process is illustrated by CH_3Hg^+ data for lake surface water in late July (Figure 4A): CH_3Hg^+ concentration is greatest at the inflow to Clay Lake and decreases progressively downstream to Ball Lake, reflecting the input of contaminated river water. In Clay Lake this pattern of variation is especially prevalent in the summer and winter (Parks, et al. 1980). In addition, CH_3Hg^+ is apparently released from lake mud into the deeper waters of the lakes, as indicated by parallels in the distribution patterns of CH Hg concentrations in bottom muds and overlying hypolimnetic waters (Figure 4A).

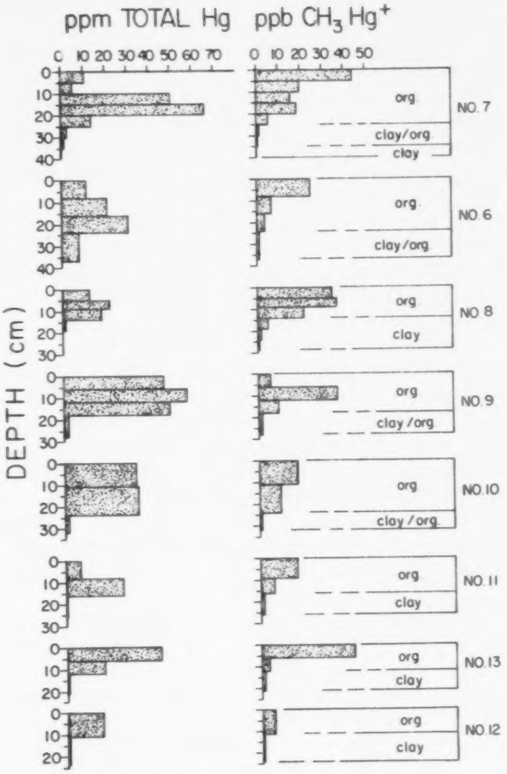

Figure 3. — Profiles of total and methyl mercury in cores from Wainwright Reservoir (no. 6-10) and the Wabigoon River upstream from it (no. 11-13). "Org." refers to wood-chip deposits overlying natural clay sediment of pre-industrial riverbed.

A crucial question from the standpoint of lake ecology and restoration is to what extent Hg accumulation by fish is due to "allochthonous" riverborne CH_3Hg^+ as opposed to "autochthonous" CH_3Hg^+ generated locally within the lakes. Comparison of our data for sediments and water with other workers' data for fish and crayfish provides some helpful clues (Figure 4, A & B). Mean Hg concentrations in different species of pelagic fish (walleye, pike, cisco, whitefish, and sauger) decrease from Clay Lake to Ball Lake (and this trend continues at least as far as Tetu Lake near the Manitoba border), paralleling the trend shown by allochthonous CH_3Hg^+ In contrast, the mean Hg concentrations of bottom-feeding animals *increase* (as in the case of suckers), paralleling the tendency shown by sedimentary CH_3Hg^+ content, or show no significant change (as in the case of crayfish). The results suggest that Hg contamination of pelagic fish (the species of particular importance to fishermen) is due primarily to CH_3Hg^+ loadings from the river, while bottom-feeding animals are contaminated to an equal or greater extent by CH_3Hg^+ generated in local bottom sediments.

The factors controlling the concentrations of CH_3Hg^+ and other Hg species in bottom sediments varied greatly down the river-lake system reflecting the observed gradient in sediment composition.

CH_3Hg^+ concentration in the putrefying wood-chip deposits between Dryden and Clay Lake is positively correlated with the abundance of methionine relative to certain other amino acids and with total N concentration (as well as the product of the N and org. C concentrations) (cf. Langley, 1973), and tends to increase with decreasing pH (Figure 5, A-C). The data suggest that methyl mercury production in this biodegradable organic medium is simply a function of the growth and metabolic activity of microorganisms which decompose organic nutrients (Langley, 1973) and employ the methionine biosynthetic pathway for methylation of Hg (Landner, 1971; Wood, 1971). The relationship with pH is consistent with the well-known preferential formation of CH_3Hg^+ with respect to $(CH_3)_2Hg$ under acidic conditions (Fagerstrom and Jernelov, 1972). CH_3Hg^+ biosynthesis is probably

Figure 4. — Variation of (A) total and methyl mercury in water and sediments and (B) total mercury in fish and crayfish, as functions of distance downstream from Dryden. Each point on each graph represents the mean of multiple replicate samples. Sediment and water data came from the present study; fish and crayfish data were furnished by Fimreite and Reynolds (1973), Bishop and Neary (1976), B. P. Neary (unpublished data, personal communication), Armstrong and Hamilton (1973), and G. McRae and A. Hamilton (unpublished data, personal communication). Symbols in Figure 4B indicate year of sample collection:
Walleye: ◆, 1970; ◇, 1975; □, 1977.
Suckers: ■, 1970; ▼, 1972.
Crayfish: O, 1971; ●, 1974; ▲, 1976.

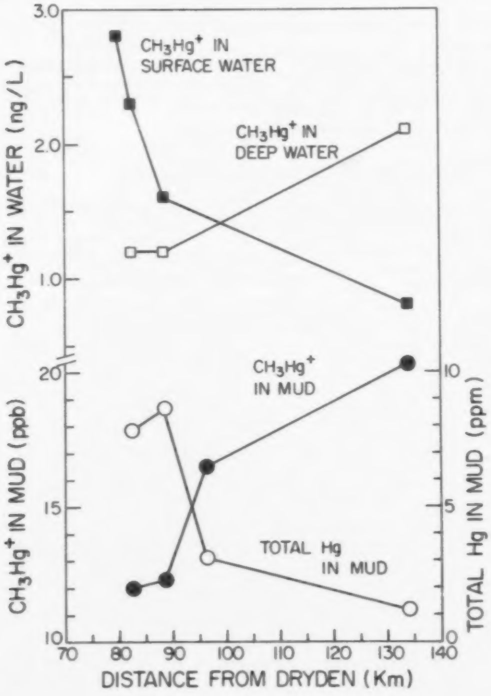

Figure 4A.

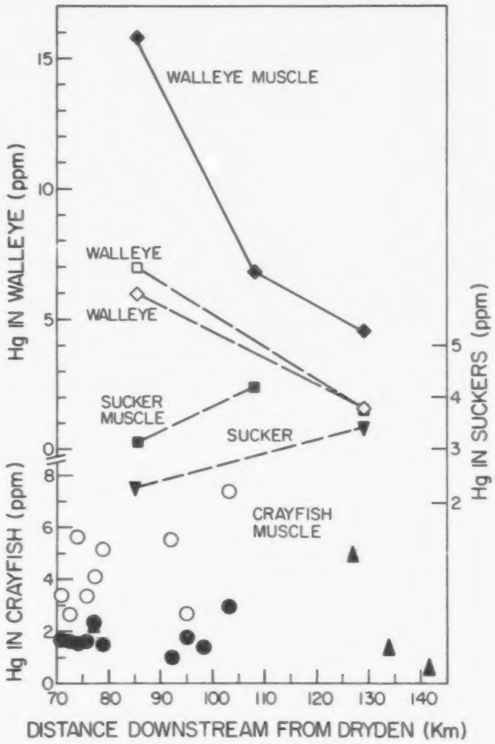

Figure 4B.

Figure 5. — Variation of methyl mercury concentration with respect to (a) methionine/threonine ratio ($r = 0.873$; $p = 0.001$-0.01) and (B) total nitrogen concentration ($r = 0.745$; $p = 0.01$-0.02) in Wainwright Reservoir woodchip sediments (cores 7 (●), 8 (■), and 10 (▲) from west side of reservoir), and (C) pH in wood-chip sediments in Wainwright Reservoir (●) and the Wabigoon River between the industrial complex and the reservoir (■) ($r = -0.596$; $p = 0.001$-0.01).

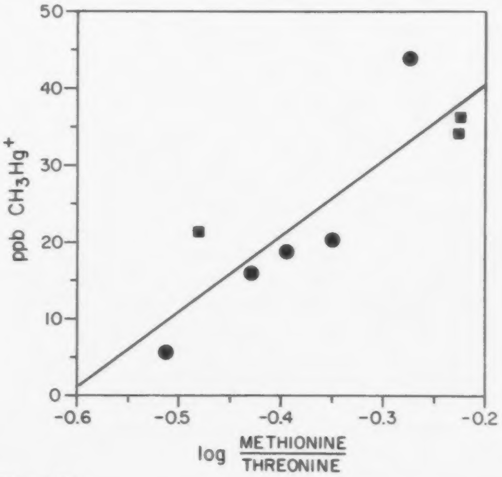

Figure 5A.

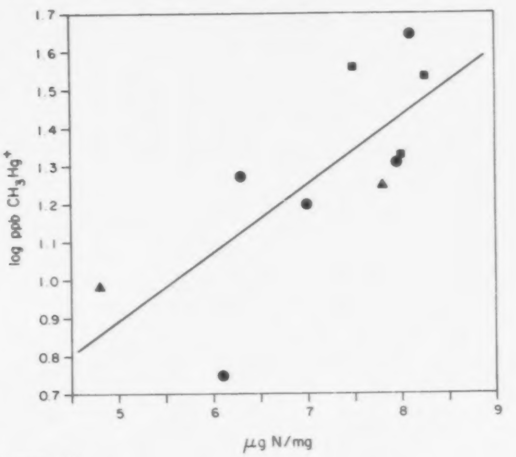

Figure 5B.

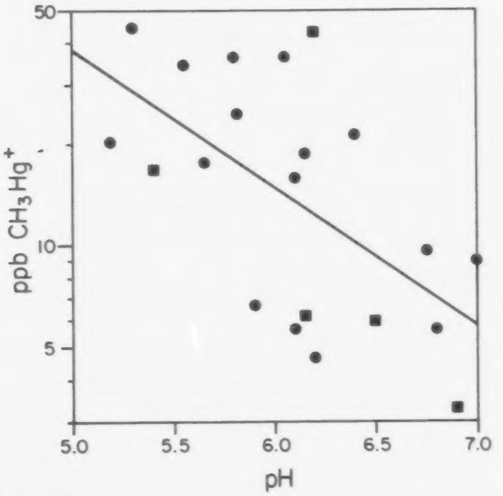

Figure 5C.

stimulated both by microbial activity *per se* and by acidic metabolic wastes resulting from it. Both are maximized at the sediment-water interface; this presumably reflects the importance of oxygen availability (cf. Vonk and Sijpesteijn, 1973), even though sediment E_h values throughout the system ranged from -420 to -50 mV, indicating anaerobic conditions.

Downstream from the inflow to Clay Lake, these effects linked directly to microbial nutrient metabolism appear to be increasingly obscured by sorption-desorption phenomena, complexation, and precipitation involving colloidal minerals, humic matter, and sulfide, Clay Lake representing a transition zone. At the east end of Clay Lake (the mouth of the inflowing river), CH_3Hg^+ levels again correlate with methionine levels but also (and to an equal degree) with the relative abundances of CBD-extractable Fe and $NH_2OH{\cdot}HCl/HNO_3$-extractable Mn (Figure 6), suggesting that the nature of the hydrated oxides affects microbial methylating activity (possibly owing to preferential fixation of Hg^{2+} ions by the more "amorphous" Mn oxide, leading to inhibition of CH_3Hg^+ production). Further downstream, the apparent role of methionine-synthesizing microorganisms and riverborne organic nutrients becomes insignificant. From the center of Clay Lake's east basin to the south basin of Ball Lake, sedimentary CH_3Hg^+ levels gave a *negative* correlation with the total or interstitial N X org. C concentration product (Figure 7), suggesting that the humified organic matter in this part of the river system, unlike the biodegradable wood chips (cf. Figure 5B), inhibits methylation — perhaps by complexing Hg ions. In both lakes, CH_3Hg^+ abundance also varies with different parameters representing the composition or physical state of the hydrated Fe and Mn oxides (Jackson and Woychuk, 1980), except in the west basin of Clay Lake, where methylation is probably limited by sulfides (Figure 2, B & C).

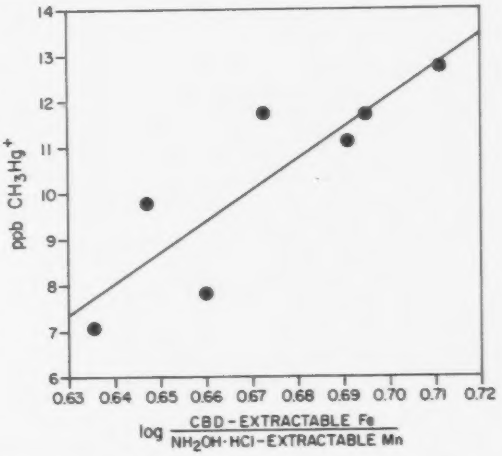

Figure 6. — Variation of methyl mercury content with the ratio of CBD-extractable iron NH OH.HCl/HNO_3 - extractable manganese ($r = 0.867$; $p = 0.01$-0.02) at the east end of Clay Lake.

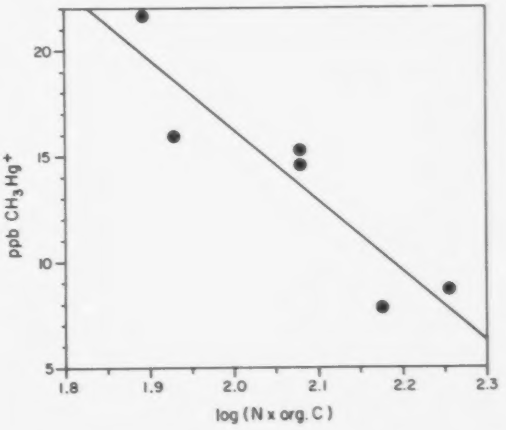

Figure 7. — Relationship between methyl mercury content and the product of the total nitrogen and organic carbon concentrations in sediments from the center of the east basin of Clay Lake ($r = -0.914$; $p = 0.001$-0.01).

Extraction of sediments with mild reagents such as 1N Ca acetate revealed a progressive increase in the abundance of the more weakly bound, or "exchangeable," Hg species relative to total Hg as the wood-chip sediments grade into clay-silt mud with increasing distance from the source of pollution (Figure 8A). This gradient in the binding characteristics of Hg may have important implications for CH Hg production. The clay-silt muds extending from Clay Lake to Ball Lake gave a strong positive correlation between mean CH Hg content and the mean ratio of Ca acetate-extractable (exchangeable) to NH OH HCl/HNO -extractable (amorphous-oxide bound) Hg species (Figure 8B). A reasonable interpretation of this relationship would be that Hg ions on exchange sites are more readily available for biomethylation than Hg ions chemisorbed, complexed, or occluded by amorphous hydrated oxides and associated humic matter.

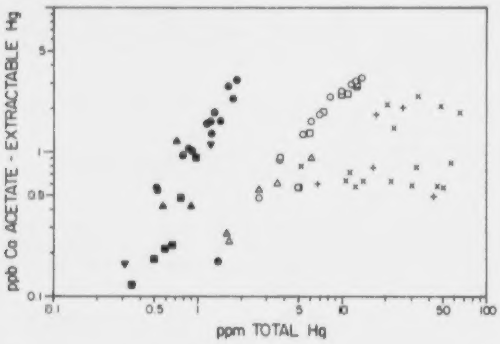

Figure 8A. — Variation of calcium acetate-extractable mercury with respect to total mercury in sediment samples from Wainwright Reservoir (wood chips, X; clay, ▼), the Wabigoon River upstream from the reservoir (wood chips, u; clay, ▲) Clay Lake (east basin, O ; west basin, □), the Wabigoon River downstream from Clay Lake (△), and Ball Lake (south basin, ● ; north basin, ■).

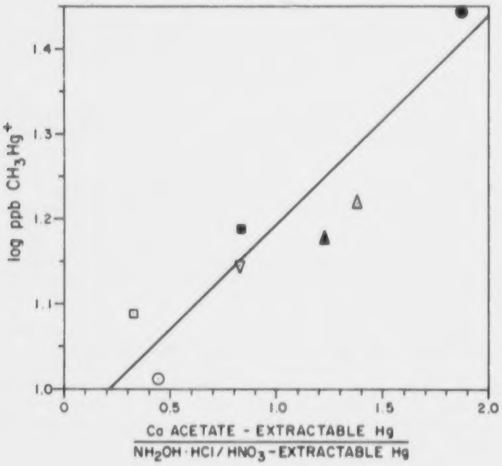

Figure 8B. — Relationship between mean methyl mercury concentration and the mean ratio of Ca acetate-extractable mercury to NH OH.HCl/HNO -extractable mercury in clay-silt bottom sediments of Clay Lake (east end of east basin, O ; center of east basin, ▽; west basin, □), the Wabigoon River downstream from Clay Lake (△), and Ball Lake (south basin: CH_3Hg^+-rich, ●; CH_3Hg^+-poor, ▲; north basin, ■)) ($r = 0.917$; $p = 0.001$-0.01).

Finally, note that CH_3Hg^+ levels are nearly the same in mud from the south basin of Ball Lake as in the wood-chip deposits near Dryden (Figure 2B) but for very different reasons. Despite relatively strong binding of Hg by the wood-chip sediments, (implying low availability for methylation), CH_3Hg^+ production near Dryden is fostered by the abundance of organic nutrients, acidity, and other conditions which stimulate growth of CH_3Hg_2-synthesizing microorganisms, as well as by the paucity of sulfide. In Ball Lake, however, CH_3Hg_2 production is fostered by the relatively weak sorption of Hg by the abundant clay and silt despite much lower concentrations of organic nutrients, higher sulfide levels, and higher pH.

Discussion: Implications for Restoration of the River-Lake System.

Any program to restore the river-lake system (that is, to lower the Hg content of its fish to background levels) would have to include (1) removal or immobilization of the sedimentary Hg in the approximately 85 km stretch of the Wabigoon River between Dryden and the inflow to Clay Lake (an estimated 2 to 3 metric tons of Hg) (Jackson and Woychuk, 1980)); and (2) prevention of further Hg discharges from the industrial complex at Dryden. Unless these steps were taken, any attempt to ameliorate the lakes individually would be doomed to failure by the constant influx of bio-available Hg from the river above Clay Lake. Halting the fluvial transport of CH_3Hg^+ and other Hg species from sources between Dryden and Clay Lake would not solve the Hg problem completely, as surface sediments far beyond this segment of the system are themselves secondary sources of bio-available Hg. Nevertheless, it could bring about a rapid, substantial decrease in the Hg levels in pelagic fish species. Such action is feasible, whereas decontamination of the surface sediments in the entire river-lake system would probably not be financially or technically practical, considering the vastness, irregularity, and poor accessibility of the system.

The following methods for ameliorating the system have been receiving serious consideration:

1. *Dredging*. Removal of contaminated sediment by dredging the riverbed between Dryden and Clay Lake could be the most effective procedure, but it would be extremely expensive and time-consuming, besides entailing the problems of dredge-spoil disposal, incomplete removal of contaminated sediment, and resuspension of fine particles. At an estimated rate of $10 to $50 /cu. yd., the project would probably cost between $40,000,000 and $200,000,000 (plus $1,000,000 for access roads) and could take up to 35 years (Wilkins, 1980; Wilkins and Irwin, 1980).

2. ***Accumulation and immobilization of Hg in a chain of holding ponds followed by eventual removal or burial***. This method (Jackson and Woychuk, 1980; Parks, et al. 1980) would require establishment of a chain of ponds by damming the river at different points between Dryden and Clay Lake. Contaminated sediments hydraulically excavated from the riverbed

(perhaps by manipulation of flow velocities) would accumulate in the ponds, and treatment with chemicals and adsorbants could be used to immobilize sedimentary Hg, scavenge Hg species from the water, and inhibit CH_3Hg_2 formation. Possibly native sulfur, an abundant and as yet unwanted byproduct of petroleum refining in western Canada, could play a useful role. Eventually the deposits in the ponds could be either dredged out or sealed off with a thick layer of sand and gravel. This method might be less expensive and more feasible than dredging the river, although a cost/-benefit analysis has not yet been done.

3. *Prevention of further Hg discharges from the industrial complex*. Release of Hg from the paper company is expected to be reduced by forthcoming (a) modernization of the plant, involving replacement of suspected sources of Hg such as old sewers; and (b) installation of a primary clarifier and retention lagoons for secondary treatment of effluent in compliance with an Ontario government control order. However, the secondary treatment (aeration and biodegradation of organic refuse) might increase the rate of CH_3Hg_2. production in the effluent.

4. *Controlled addition of selenium compounds to the river system*. This ingenious but controversial approach is still in the experimental stage (Rudd, et al. 1980). Possible toxic effects of the selenium would have to be investigated exhaustively before the method could be recommended.

Additional restoration techniques were examined but were judged to be unsuitable. These included (a) burial of polluted sediments with uncontaminated clay and silt (Rudd, et al. 1980; Wilkins and Irwin, 1980); (b) ploughing of lake bottom sediments (Wilkins, 1980); and (c) diversion of the Wabigoon River from its present channel (Wilkins and Irwin, 1980).

The option of simply leaving the river-lake system alone was also considered. Hg levels in fish have been declining steadily since 1970, when uncontrolled discharges ceased; but the slowness of the decline and the presence of huge Hg accumulations undergoing methylation in the riverbed above Clay Lake lead to the conclusion that restoration by natural processes would take many decades, if not centuries (Jackson and Woychuk, 1980; Parks, et al. 1980).

In conclusion, there is hope that the river-lake system can be ameliorated, although any program for achieving this would have some limitations and uncertainties. From the standpoint of ending the human misery caused by the Hg pollution, the most satisfactory solution might be resettlement of the Grassy Narrows and White Dog communities.

The poisoning of the Wabigoon-English-Winnipeg River system demonstrates the urgent need for rigorous enforcement of strict legislation banning toxic substances from effluents. Clearly, prevention is the best cure.

ACKNOWLEDGEMENTS

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SIMPLIFIED ECOSYSTEM MODELING FOR ASSESSING ALTERNATIVE BIOMANIPULATION STRATEGIES

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ABSTRACT

The technique of "loop analysis" was applied to a variety of hypothetical lake ecosystems in an attempt to assess qualitatively the potential of biomanipulation as a lake restoration technique. Loop analysis is particularly well suited for evaluating complex ecosystems, where in many cases system interactions can be specified only qualitatively. The technique is based on the equivalence of a set of linear differential equations at or near equilibrium and their matrices and loop diagrams. The only information required is that the number of system components and their direct interactions (in terms of positive, negative, or no impact) be specified. In general, results support current ecosystem theory and several recent field studies. In simple chain systems, perturbations typically produce alternating effects, with perturbations from the top having more impact than perturbations from the bottom. Similarly, planktivorous fish appear to play an important role in structuring lower trophic levels. However, in more complex food webs, results are not nearly so predictable, and in some cases are completely opposite to what might be expected. For example, in certain systems an increase in the density of game fish leads to an eventual decrease in the game fish; however, algae also experience a decrease. On the other hand, a decrease in game fish leads to an increase in algae, a result which may have serious fisheries management implications. It is conceivable that, in some lakes, changes in water quality may be directly attributable to fishing pressure rather than to a change in nutrient status.

INTRODUCTION

Biomanipulation as a potential lake restoration technique has been discussed for a number of years. In particular, Patten (1973) and Shapiro, et al. (1975) pointed out the relative ease with which upper trophic levels could be manipulated to control lower levels. Recently, more and more experimental evidence is surfacing to support the contention that biomanipulation may be a viable lake restoration technique (Andersson, et al. 1978; Briand and McCauley, 1978; Gliwicz, 1975; Haertel, 1977; Henrikson, et al. 1980; LeBrasseur, et al. 1978; Lynch, 1979; Molotkov, et al. 1978; Porter, 1977; Roman, 1978; Smyly, 1978; von Ende, 1979). In fact, in some situations it may be the only feasible technique.

Biomanipulation certainly is not a new concept; it has been a standard tool of fisheries management for many years. Understandably, the emphasis has been placed almost exclusively on the upper trophic level fisheries. Unfortunately, there has been little regard for the rest of the ecosystem. What is presently needed is a more holistic ecosystem management approach to direct system productivity not only toward desirable upper level components but also away from undesirable components such as algae.

In the past, many lake restoration activities have focused on nutrient abatement using the simple input/output models of Dillon and Rigler (1974) and Vollenweider (1975, 1976) to predict resulting water quality improvements. Unfortunately, no similar technique has been developed to assess potential impacts of various biomanipulation strategies; as a result, ecosystem management to improve water quality is essentially untested on a whole-lake scale, and application will likely remain limited to isolated case studies until a suitable model is developed. The purpose of this paper is to present a simple modeling technique called "loop analysis," which may satisfy some of these needs.

MODEL DESCRIPTION

Loop analysis was developed by Levins (1974, 1975) to qualitatively evaluate ecosystem stability and interaction. The technique has been applied to both terrestrial and aquatic ecosystems (Levins, 1975), model plankton communities (Lane and Levins, 1977), and a hypothetical aquatic food web (Briand and McCauley, 1978). This paper extends past work by examining a variety of hypothetical biomanipulation strategies.

Mathematically, loop analysis is founded in matrix algebra and is based on the equivalence of linear differential equations at or near equilibrium. The equations are of the form

$$\frac{dX_i}{dt} = F_i(X_1, X_2, X_1, \ldots X_1,)$$

where x's are the system variables. Variables are usually species or trophic level abundance, but can also be nutrient levels or even such factors as toxic byproducts or predation pressures. At or near equilibrium, the behavior of the system depends on the properties of the community matrix.

$$A = \begin{array}{ccccc} a_{11} & a_{12} & a_{13} & \cdots & a_{1n} \\ a_{11} & a_{12} & a_{13} & \cdots & a_{1n} \\ a_{11} & a_{12} & a_{13} & \cdots & a_{1n} \\ \cdot & \cdot & \cdot & \cdots & \cdot \\ \cdot & \cdot & \cdot & \cdots & \cdot \\ \cdot & \cdot & \cdot & \cdots & \cdot \\ \cdot & \cdot & \cdot & \cdots & \cdot \\ a_{n1} & a_{n2} & a_{n3} & \cdots & a_{nn} \end{array}$$

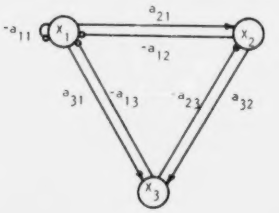
where the elements a_{ij} are given by

$$a_{ij} = \frac{\partial F_i}{\partial X_i}$$

and are simply the coefficients of the X_i in Eq. (1) for dx_i/dt, evaluated at equilibrium. For example, a_{11} is the coefficient used to describe the impact of X_1 upon itself; the coefficient a describes the impact of X_2 on X_1, etc. The technique of loop analysis is basically a shorthand method for solving the community matrix.

Schematically the systems may be represented by directed diagrams, as in Figure 1. Here the variables X_1, X_2, and X_3 are the vertices of the diagram, and the interactions between the variables are represented by oriented links. In most ecosystems, the actual values of the matrix coefficients are not known; however, the direction and signs of interactions can usually be specified. In the above system, arrows indicate positive links and circles are negative links. For example, X_1 might be an algal species, X_2 an herbivorous zooplanktore, and X_3 an omnivorous zooplanktore. Thus X_2 feeds upon X_1, and X_3 preys upon both X_1 and X_2.

It should also be noted that variables may affect themselves, either positively or negatively. In Figure 1, X_1 has a negative impact on itself which is termed self-damping. In general, all resources which are not self-reproducing, such as mineral nutrients, organic matter, or detritus, are self-damped. If the resource is not specifically included in the diagram, then the organism which requires that resource incorporates the self-damping, as in Figure 1. Thus, all ecosystems are limited or self-damped by their ultimate food source.



$$A = \begin{bmatrix} -a_{11} & -a_{12} & -a_{13} \\ a_{21} & 0 & -a_{23} \\ a_{31} & a_{32} & 0 \end{bmatrix}$$

Figure 1. — Loop diagram and community matrix of a three-variable ecosystem.

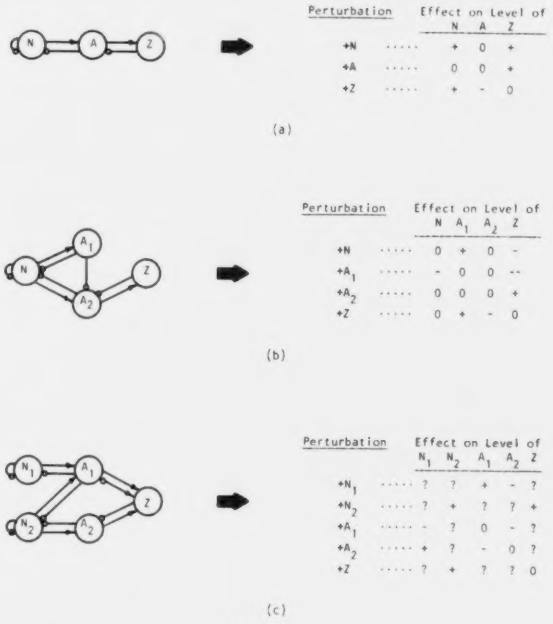
Because the theoretical development and methodology required for actually solving loop analysis problems is complicated and quite lengthy, this author will not attempt to elaborate on it in this paper. Rather, the following discussion will focus on information required to perform loop analysis, application of the technique to selected lake ecosystems, and potential significance and limitations of results. For a formal treatment of loop analysis theory and methodology, the interested reader should consult Levins (1975).

The information necessary to perform loop analysis is minimal; all that is required is that variables be selected which adequately describe the system, and that the direct variable interactions be specified qualitatively. For example, Figure 2a depicts a system containing a nutrient, an algal species, and zooplankton species. It can be noted by the directed links between variables that the nutrient is self-damped and that it benefits the algae. The algae deplete the nutrient but enhance zooplankton. The zooplankton feed on the algae, as indicated by the negative link. It is important to note at this point that the diagrams are not descriptive of whole ecosystems. Numerous other inputs and outputs may be involved (sunlight, temperature, losses due to non-predatory mortality); these are not shown. The diagrams attempt to portray only system variables; all other components are assumed to be constant, or their rates of reaction are constant.

In Figure 2b, two species of algae compete for a single nutrient. Zooplankton feed on algae A_2, but A have no direct impact on A_1.

Figure 2c is a bit more complicated. There are two nutrients, two algal species, and a single zooplankton species. Algae A_1 require both nutrients, but A_2 need only N_2. Zooplankton feed on both algal species.

The sample diagrams are not intended to represent real lake ecosystems; rather, they are meant to illustrate the flexibility of diagram construction. It is



Perturbation		Effect on Level of		
		N	A	Z
+N	.....	+	0	+
+A	.....	0	0	+
+Z	.....	+	-	0

(a)

Perturbation		Effect on Level of			
		N	A_1	A_2	Z
+N	.....	0	+	0	-
$+A_1$	.....	-	0	0	--
$+A_2$	.....	0	0	0	+
+Z	.....	0	+	-	0

(b)

Perturbation		Effect on Level of				
		N_1	N_2	A_1	A_2	Z
$+N_1$	.....	?	?	+	-	?
$+N_2$	.....	?	+	?	?	+
$+A_1$	.....	-	?	0	-	?
$+A_2$	.....	+	?	-	0	?
+Z	.....	?	+	?	?	0

(c)

Figure 2. — Loop diagrams and perturbation analyses of partially specified lake ecosystems.

quite easy to see that the range of combinations and permutations of different variables and interactions is limited only by imagination.

The formal analysis of loop diagrams may have several end results, but for this paper only that aspect which deals with the effects of biomanipulation will be emphasized. In particular, we will examine how a system assumed to be at or near equilibrium adjusts to a new equilibrium as a result of changes imposed on the system. These changes or perturbations may be step changes, such as an increase in phosphorus input, or evolutionary changes which enhance a biological variable's ability to survive in the system. Recently the technique has been expanded to include transient and periodic perturbations (Flake, 1980), but this will not be considered in this paper.

In Figure 2, perturbation analyses are illustrated for sample systems. The specific perturbation is indicated by a +N, +A, etc., and the predicted effect of each perturbation on the levels of all system variables is shown by the matrix-type diagram*. To avoid redundancy, only positive perturbations are illustrated; the effects of negative changes can be derived simply by reversing the signs.

In Figure 2a, one would predict that an increase in the nutrient supply would increase both the nutrient and the zooplankton levels, but would not affect the level of algae in the system. However, it is important to note that while the level of algae is unaffected, its rates of interaction with other variables may change. Since the level of nutrient has increased, algae would presumably respond with an increased growth rate. However, zooplankton predation on algae increases because zooplankton themselves increase. Thus, the increase in algae productivity is passed directly to the zooplankton without increasing algal abundance.

The second perturbation in Figure 2a is that of enhancing the algae component. This results in increasing zooplankton abundance with no changes in either the nutrient or the algae. Finally, the enhancement of zooplankton results in an increase in nutrient levels, a decrease in algal biomass, and no change in zooplankton.

In Figure 2b, results of a similar nature are portrayed. It can be seen that nutrient levels are affected only by perturbations of A_1 and that levels of A_2 are affected only by changes in zooplankton. Another item of interest is the effect of changes in A_1 on zooplankton levels. The double negative sign indicates that two viable pathways through which A_1 may react with zooplankton, both of which have negative impact. It is important to remember, however, that since this technique is entirely qualitative a double negative does not necessarily have any more quantitative significance than does a single negative. Nevertheless, it is interesting qualitatively, and for that reason it is included in the results.

Finally, in Figure 2c it may be noted that the matrix is largely filled with question marks. This results when a perturbation affects a variable through two or more viable pathways, the signs of which are opposite. Because the magnitude of the impacts is unknown, the net effect cannot be resolved; thus, the results are ambiguous. Increases in system complexity — particularly in foodweb situations — quite often lead to increases in the number of ambiguous results. This factor very possibly could be the most critical limitation in applying loop analysis to complex ecosystems.

APPLICATION

In the following pages, the author will present three hypothetical lake ecosystems and perturbation analyses. The attempt is to portray a range of "real" systems with the ultimate objective of reducing algae levels through manipulation of other biological components. Biological components are specified by functional group rather than at the species level in an attempt to avoid unnecessary duplication. Thus, for example, the forage fish component may be composed of many species of minnows, smelts, and young game fish. Similarly, algae may be grouped by size and edibility instead of by the more traditional green/blue — green/diatoms grouping. The intent is that items in a group act and react similarly, irrespective of their specific taxonomies.

In Figure 3a, a simple chain system is illustrated which contains phosphorus (P), algae (A), zooplankton (Z), forage fish (Sm) (predominately smelts), and game fish (Sa) (predominately landlocked salmon). The coldwater fishery was selected because of its significance to Maine. A similar diagram may describe many warmwater fisheries. It should be pointed out that in this system — and in those following — the game fish are considered to be self-damping. In this case, self-damping is included to account for the influence of man on the ecosystem. Most lakes support sport fisheries which have a direct negative impact on the game fish and an indirect impact on other system components. The end result is that the game fish level is no longer totally responsive to the rest of the system; the population is damped by man's influence, regardless of what the rest of the system is doing.

Results of the perturbation analysis are illustrated in Figure 3a. An increase in phosphorus input leads to an increase in the levels of all system components, which corresponds with many field observations and also with common sense. Within limits, all biological components should benefit from an increase in fertility.

Increases in the other components produce variable responses. For example, an enhancement of zooplankton reduces algae, increases phosphorus, and increases smelt and salmon levels. In general, it can be seen that components below the perturbed level respond with alternating effects, while those above experience identical responses. It is also interesting to note that the highest trophic level in the chain responds to perturbations in all levels, and vice versa, perturbations in the highest level result in changes in all other levels. That upper trophic level components should inherently have considerable influence over ecosystem structure and response has been stated previously (Patten, 1973).

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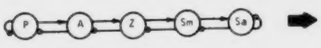
* This matrix is in no way related to the community matrix described previously. It is simply a convenient format for portraying results.

Finally, the impact of perturbations on the algae component can be examined. It can be seen that enhancement of zooplankton and salmon should produce decreases in algae levels, and increases in the phosphorus and smelt components. Enhancement of the algae itself produces no change in algae, because of a reduction of its own food supply and an increase in predation.

The results presented in the previous example are for the most part not surprising; the structure and dynamics of simple food chains have been known for some time. However, minor aberrations in the chain structure may have drastic impacts on system dynamics, and in more complex food webs many results are not at all predictable.

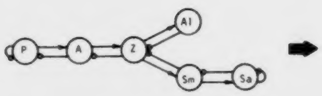
The system illustrated in Figure 3a probably represents the past ecosystem for Echo Lake, Maine. Salmon production was not satisfactory, and in an attempt to improve productivity, landlocked alewives were introduced as an additional forage source. This was a reasonable move from a fisheries management perspective. From Figure 3a, loop analysis would predict that enhancing the forage fish should increase salmon biomass. Unfortunately, to date the salmon refused to feed significantly on the alewives.

The modified ecosystem is illustrated in Figure 3b. Schematically, the only change in the system is the addition of the alewife component connected with the zooplankton component; however, the results of the perturbation analysis dramatically differ from the previous example. The most obvious difference is the lack of response of most levels to most perturbations. In particular, increasing phosphorus input increases algae and alewives with no changes in the other components. Similarly, enhancement of the salmon increases alewives, decreases smelts, but produces no other changes. However, enhancement of the alewife component decreases all other components except algae, which increase, and alewives, which remain the same. By the same token, alewives respond positively to enhancement of all other components except smelt, which decrease alewives because they compete directly for food.



Perturbation	Effect on Level of P	A	Z	Sm	Sa
+P ······	+	+	+	+	+
+A ······	-	0	+	+	+
+Z ······	+	-	0	+	+
+Sm ······	-	+	-	0	+
+Sa ······	+	-	+	-	+

(a)

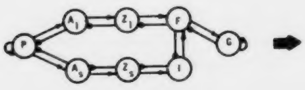


Perturbation	Effect on Level of P	A	Z	Al	Sm	Sa
+P ······	0	+	0	+	0	0
+A ······	-	0	0	+	0	0
+Z ······	0	0	0	+	0	0
+Al ······	-	+	-	0	-	-
+Sm ······	0	0	0	-	0	+
+Sa ······	0	0	0	+	-	0

(b)

Figure 3. — Loop diagrams and perturbation analyses of past (a) and present (b) simplified ecosystems for Echo Lake, Maine.

Interpretation of these results indicates that introduction of alewives into Echo Lake has effectively short-circuited the existing food chain, benefiting the alewives and, importantly, also algae. Furthermore, since alewives are neither harvested by man nor preyed upon by game fish in this system, they are free to act as a system buffer, readily absorbing changes while other components — with the possible exception of algae — remain largely unaffected. Fortunately, Echo Lake is oligotrophic, and nuisance algae conditions have not beome a problem. Nevertheless, a large and unutilized alewife population presently exists; salmon productivity has not improved.



Perturbation	Effect on Level of P	A_l	A_s	Z_l	Z_s	F	I	G
+P ······	+	+	?	+	+	+	?	+
$+A_l$ ······	+	0	--	++	+	+	--	+
$+A_s$ ······	--	?	0	--	?	?	++	?
$+Z_l$ ······	+	?	?	0	+	+	?	+
$+Z_s$ ······	+	+	--	+	0	+	?	+
+F ······	+	+	--	+	+	0	--	+
+I ······	--	?	++	--	--	?	0	?
+G ······	-	-	++	-	-	-	++	?

Figure 4. — Loop diagram and perturbation analysis of the eight-variable ecosystem for Hermon Pond, Maine.

The final system, presented in Figure 4, is at least in part representative of another Maine lake, Hermon Pond. (Ecosystem data are still being collected.) System components include a nutrient source, large and small algae, large and small herbivorous zooplankton, an aquatic insect group, forage fish, and game fish. The insect level is composed primarily of *Chaorborus* species, with minor amounts of predaceous zooplankton. The forage fish level is predominately smelt, yellow perch, and small white perch. The game fish are mainly large white perch, smallmouth bass, and a few chain pickerel. The large algae component contains most of the undesirable species and is therefore the level to be minimized.

Results of the perturbation analysis are mixed — some responses are similar to those for chain systems, and others are quite different. Conflicting pathways have produced ambiguous results, but in general they do not appear to limit seriously the utility of the analysis. As in the chain system, an increase in phosphorus input increases most other components. However, enhancement of the game fish decreases all components except small algae and insects; as illustrated by this system, a general conclusion is that the alternating impacts apparent in chain systems are invalid in web systems.

It may be noted that only one perturbation produces an unambiguous negative effect on the large algae — that of enhancing game fish. Even increasing large zooplankton does not definitely reduce large algae. This is because the impact of the direct path — large zooplankton to large algae — is at least in part negated by the impact of the indirect path — large zooplankton-forage fish-insects-small zooplankton-small algae-phosphorus-large algae. Thus, it appears that a management strategy for this pond might be to increase game fish, perhaps through stocking or through harvest reduction (or to decrease forage fish). Though a decrease in small zooplankton should also have a negative impact on large algae, man's ability to directly manipulate zooplankton levels is probably quite minimal.

DISCUSSION AND SUMMARY

For biomanipulation to become a reality in lake restoration efforts, a technique must be available to predict the entire community response to that manipulation. As a first-cut qualitative approach, loop analysis satisfies some of these needs. The preceding examples demonstrate the generality and versatility of the technique. The variety of ecosystem structures and interactions that can be examined is in theory infinite, and application should be possible for almost any aquatic ecosystem imaginable. Results from the perturbation analyses often support ecological theory and field observations, but are sometimes surprising in that some are entirely opposite to what one would intuitively expect. It is in these counter-intuitive results that the real strength of loop analysis lies — in firm partnership with biological knowledge, the insight gained from loop analysis can help unravel complex system interactions which have long frustrated ecological researchers. And when a holistic approach is adopted, it becomes apparent that even results which seem unreasonable on the surface are in fact quite reasonable when the system as a whole is considered.

Loop analysis is not, however, without faults. As with any mathematical model, underlying assumptions must be kept foremost in mind to prevent misusing the technique. First, and perhaps most important, it is assumed that ecosystems can be described by linear equations. In fact, linearity in real systems is probably the exception rather than the rule. Nevertheless, the analysis attempts only to identify qualitative trends in ecosystem dynamics — the magnitude of those trends cannot be deduced (nor should it be inferred). In this light, it is assumed that a linear approximation is adequate.

Second, it is assumed that the ecosystem components and interactions have been adquately described. It has been shown that seemingly minor changes in system structure may result in major changes in system response to perturbations. In applying loop analysis to real lake systems, it becomes imperative that the structure of the specific ecosystem be known in detail.

Third, it is assumed that the systems are in equilibrium or at least in moving equilibrium. While this assumption is common with many types of models, it still warrants emphasis since many culturally eutrophic lakes are probably not in equilibrium, particularly ecologically.

Ultimately, improvement in managing community structure in lake ecosystems will result primarily from field experimentation rather than from mathematical modeling. Lake systems are too complex to expect otherwise. However, field experimentation, unless guided by theoretical knowledge of subsystem interactions, involves testing a multitude of management possibilities and sorting out their ecosystem impacts from the effects of all uncontrolled, natural variations occurring simultaneously. Loop analysis can play a much-needed intermediate role by providing the insight necessary to guide future experimentation.

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RESPONSE OF ZOOPLANKTON IN PRECAMBRIAN SHIELD LAKES TO WHOLE-LAKE CHEMICAL MODIFICATIONS CAUSING pH CHANGE

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ABSTRACT

Two lakes, 227 and 223, in the Experimental Lakes Area of northwestern Ontario, have been subjected, respectively, to whole-lake addition of fertilizer, nitrogen and phosphorus, and to addition of sulfuric acid. Effects on their zooplankton populations are believed to be brought about largely by changes in the pH. The low endogenous concentrations of dissolved inorganic carbon render these lakes prone to extreme pH change. Phosphorus input to lake 227 was increased 10-fold each year from 1969 to 1974 by adding N:P in 15:1 ratio. Maximum mid-summer biomass of cladocerans and calanoids declined each year after fertilization reaching very low levels by 1972. Cyclopoid biomass was only moderately reduced following fertilization. Rotifer biomass increased manyfold in 1970 and 1971 but declined to very low levels by 1974. Mid-summer epilimnion pH levels recorded were frequently above 10.0. Enhanced decomposition caused the anoxic zone on the lake bottom to deepen significantly. Changes in crustacean biomass are thought to be due to loss of oxygenated, near-neutral habitat within the water column. A relationship between rate of phosphorus loading, endogenous dissolved inorganic carbon, and epilimnion pH from 6.7 to 5.84 and severely reduced the dissolved inorganic carbon. By 1979 neither total zooplankton biomass nor species diversity has changed appreciably. Nevertheless, the species composition changed somewhat. Rotifers showed various responses. No predictive relationships were evident between species' tolerance of high or low pH.

INTRODUCTION

The Experimental Lakes Area (ELA) in the Precambrian Shield of Ontario was selected in the 1960's for whole-lake experimental study of eutrophication. One reason these lakes were suitable was the low conductivity of the waters, making them amenable to change to new chemical states by controlled additions of substances (Johnson and Vallentyne, 1971). The lakes are poorly buffered and thus are susceptible to changes in pH. Alkalinity results principally from the bicarbonate system derived by carbonic acid weathering of the alumino-silicate bedrock (Brunskill, et al. 1971). Bicarbonate concentration in ELA lake surface waters averages 4.1 mg HCO_3^- l^{-1} (67 μ moles l^{-1}), among the lowest reported in the world, ranking with lakes in the Adirondack Mountains, N.Y., the Cairngorm area, Scotland, (Armstrong and Schindler, 1971), and parts of Scandinavia (Wright, et al. 1976).

Changes in pH of ELA lakes have been brought about experimentally in two ways. Adding phosphate and nitrate fertilizer caused pH to increase. Enhanced photosynthesis in Lake 227 drained the total epilimnion CO_2 pool to such an extent that bicarbonate was negligible. The hydroxyl ion generated in CO_2 and nitrogen uptake by plankton dominated the alkalinity, so that pH of 10.0 or higher was frequently recorded in the epilimnion. In another experiment, the pH of Lake 223 was reduced by adding sulfuric acid. pH was lowered 0.25 to 0.50 pH units per year to quantify the rate of acidification and the biological and chemical effects resulting from the addition of known amounts of acid (Schindler, et al. 1980).

FERTILIZATION OF LAKE 227

Lake 227 is a small, oligotrophic lake, 5.0 hectares in area, with a mean depth of 4.4 m and maximum depth of 10.0 m (Schindler, et al. 1971). It was selected in 1969 for its unusually low levels of dissolved inorganic carbon (DIC) (70 μ moles l^{-1} average in epilimnion in 1969 prior to fertilization) to test whether carbon shortage limited eutrophication.

In 1969 the lake received 0.34 g m^{-2} of P as Na^2HPO^4 and 5.04 g m^{2} of N as $NaNO_3$ in 17 equal weekly additions starting in late June. During May to October of each year from 1970 to 1974, 21 weekly additions were make of P as H_3PO_4 and N as $NaNO_3$ for an annual total of 0.48 g m^{-2} and 6.29 g m^{-2} N, and a N:P ratio of 13:1 by weight. The fertilization regime was changed in 1975 to 1978 so that N:P ratio was about 5:0 by weight. During the latter years 20 weekly additions were make for annual loadings of 0.46 g m^{-2}P as H_3PO_4 and 2.25 g m^{-2}N as $NaNO_3$. The lake, methods of fertilization, and the changes in chemistry and phytoplankton following fertilization are described by Schindler, et al. 1971; 1972; 1973; Schindler and Fee, 1974; Findlay and Kling, 1975; Schindler, 1975; 1977; and Findlay, 1978.

RESPONSE OF LAKE 227 TO FERTILIZATION: 1969-1974 PERIOD

Both primary production and standing algal biomass increased in Lake 227 following the addition of P and N. Algal biomass increased several-fold after fertilization in 1969 (Schindler, et al. 1971) and reached a maximum of about 20 times the pre-fertilization values in late July 1972. Algal species composition during the ice-free season changed with addition of N and P in 13:1 ratio; chlorophytes and cyanophytes replaced cryptophytes and chrysophytes as dominants (Schindler, et al. 1973). Edible species of algae were abundantly available to the zooplankton during 1969 to 1974 (Kling, pers. comm.).

The increased primary production in Lake 227 was associated with mid- or late morning pH values above 10.0 in the epilimnion on a number of sampling dates in 1970, 1972, and 1973 (Schindler, et al. 1973). pH fluctuated little diurnally (Schindler and Fee, 1973). Photosynthetic activity draws from the free CO^2 pool which in turn is replenished from carbonate alkalinity or by invasion of CO^2 from the atmosphere. CO_2 invasion was not sufficiently rapid to supply all the required CO_2 for photosynthesis on sunny mid-summer days in Lake 227 (Schindler and Fee, 1973). The removal of CO_2 by the algae was sufficiently great to drain the free-CO_2 pool and to elevate the pH reactions such as:

$$HCO_3^- \rightleftharpoons CO_2 + OH^-$$

(King, 1970).

The anoxic bottom layer thickened in the years following fertilization reaching a maximum thickness in 1972. In mid-July 1972 several extreme conditions occurred together. The epilimnion, 0 to 2 m, was at pH above 10.0, but was well-oxygenated. Below 2 m, pH dropped to about 7, but O_2 at 3 m and below was less than 2 mg l^{-1}.

RESPONSE OF ZOOPLANKTON IN LAKE 227

Seasonal changes in abundance of species of crustaceans and rotifers in Lake 227 from 1969 to 1974 are described by Malley, et al. (In prep. a). Average number of individuals of crustacean species during May to September in a column of water under 1 m^2 of lake surface at the center of the lake for 1969 to 1978 are reported in Tables 1 to 3. Typically, the epilimnion, metalimnion, and hypolimnion were sampled separately and the numbers in each stratum per m^2 were weighted according to the volume of the stratum and summed together to give the number per m^2. Data for 1975 and 1976 are omitted for the tables because only the uppermost 2 m of the water column were sampled in those years. Total biomass of groups of zooplankton including rotifers is shown in Figure 1 for 1969 to 1974. Dry weight biomass for individuals of each species was calculated from simple geometric shapes approximating the size and shape of each life stage or size category (Lawrence, et al. In prep). Zooplankton sampling methods are described by Chang, et al. (1980).

Populations of the cladocerans *Bosmina longirostris, Diaphanosoma brachyurum, Daphnia retrocurva*, and *Holopedium gibberum* declined on the average with fertilization in the summers of 1970 and 1971 compared with abundances in the first year of fertilization, 1969 (Table 1). All four species were severely reduced in numbers or not recorded at all in 1972 and 1973. All were recorded again in 1974 but mostly at low densities except for an unusually high density of *D. brachyurum* on one date. By 1978 numbers of *D. brachyurum* were very similar to those in 1969. *B. longirostris* was abundant on one sampling date in 1978, resulting in a seasonal average as high as in 1969, but the seasonal pattern was very different. In 1969 the species was well represented throughout May to September but in 1978 was recorded only during June and July. *D. retrocurva* and *H. gibberum* failed to recover by 1978 to densities seen in 1969. Reflecting these abundances, total dry weight biomass of cladocerans was lower in 1970 and 1971 than in 1969 and very low in 1972 and 1973 (Figure 1). The large biomass on one date in 1974 is due to the high density of *D. brachyurum*. Rare species of cladocerans in the Lake 227 samples included *Ceriodaphnia* sp., *Chydorus sphaericus*, and *Alona* sp. (Chang, et al. 1980).

The calanoids, originally dominated by *Diaptomus minutus*, declined in population sizes of adults, nauplii, and copepodids from 1969 to 1972. Minimum number were found from August 1972 through 1973. Populations increased slightly in 1974 and after, but by 1978 were far below 1969 abundances (Table 2). *Epischura lacustris* was the dominant calanoid in Lake 227 in 1974, 1977, and 1978. *Diaptomus leptopus* was found occasionally in Lake 227. Total dry weight biomass of calanoids declined to a minimum after July 1972 and remained low throughout 1973 and 1974 (Figure 1).

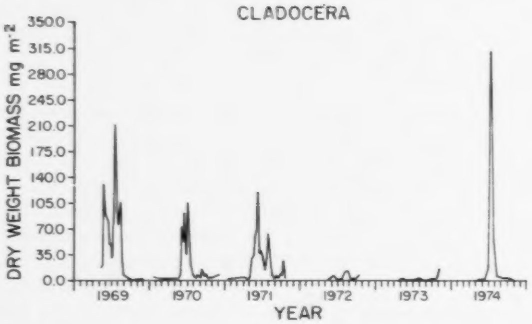


Figure 1. — Dry weight biomass (mg m^{-2}) of four groups of zooplankton in L227 during 1969 to 1974.

Cyclopoids were represented by three species (Table 3) which shifted in dominance from 1969 to 1971. *M. edax* dominated in summer 1969; *T. prasinus* in 1970, and *C. bicuspidatus* in 1971. Each species was present in low numbers or not recorded in 1972, 1973, and 1974 except for *M. edax*. All four species were present in 1977 and 1978. Populations of cyclopoid nauplii and copepodids did not decline from 1969 to 1971, but

Table 1. — Number m^{-2} of cladocerans in Lake 227 averaged over the May to September period during the years 1969 to 1974 and 1977 and 1978.

Species	1969	1970	1971	1972	1973	1974	1977	1978
Bosmina longirostris	34,450	19,150	24,350	1,800	350	4,050	450	37,900
Diaphanosoma brachyurum	8,400	3,750	3,700	700	500	22,650	4,450	11,900
Daphnia retrocurva	4,100	1,600	1,800	200	0	700	0	1,000
Holopedium gibberum	2,800	1,700	800	0	0	150	0	0

Table 2. — Number m^{-2} of calanoid ccpepods in Lake 227 averaged over the May to September period during the years 1969 to 1974 and 1977 and 1978.

Species	1969	1970	1971	1972	1973	1974	1977	1978
Diaptomus minutus adults	32,650	16,400	10,650	2,800	25	0	0	7
Epischura lacustris adults	0	0	0	0	0	150	1,250	1,050
Calanoid nauplii N_1-N_6	61,400	66,300	72,750	22,000	1,750	1,950	6,500	11,400
Caalanoid copepodids C_1-C_5	110,500	57,450	25,850	2,350	0	1,250	2,300	4,800

Table 3. — Number m^{-2} of cyclopoid copepods in Lake 227 averaged over the May to September period during the years 1969 to 1974 and 1977 and 1978.

Species	1969	1970	1971	1972	1973	1974	1977	1978
Cyclops bicuspidatus thom. adults	0	50	2,000	100	50	0	300	350
Mesocylops edax adults	2,550	350	400	100	0	550	550	950
Tropocyclops prasinus mex. adults	1,700	700	0	0	0	0	150	100
Cyclopoid nauplii N_1-N_6	55,600	95,550	109,900	45,200	49,200	57,850	16,650	42,700
Cyclopoid copepodids C_1-C_5	59,750	16,600	54,350	14,150	9,150	21,800	4,000	15,100

were smaller in 1972 and 1973. Overall, cyclopoids did not decline in numbers with fertilization as dramatically as did calanoids and cladocerans. This is illustrated further for dry weight biomass in Figure 1. Nevertheless, during the 1969 to 1974 period standing cyclopoid biomass was at a (summer) minimum in August 1972.

Individual species of rotifers responded variously to fertilization from 1969 to 1974. Nevertheless, numbers of a species were generally higher in 1970 and 1971 than during 1969 and most species were reduced in 1974 to below their abundances in 1969. These results are shown in Figure 1 for total rotifer dry weight biomass. The increases in biomass in 1970 and 1971 reflect larger population sizes of *Keratella cochlearis*, *Polyarthra vulgaris*, and *Anuraeopsis fissa*.

CAUSES OF ZOOPLANKTON DECLINE IN LAKE 227

The decline of crustacean zooplankton, particularly herbivores, with fertilization of Lake 227 was an unexpected result. Eutrophication usually increases the standing crop of herbivores (Brooks, 1969; Smith, 1969; Hillbricht-Ilkowska and Weglendska, 1970; LeBrasseur and Kennedy (1972) or produces no change (Briand and McCauley, 1978). The most plausible explanation for the decline of the zooplankton in Lake 227 is that the combination of high pH in the epilimnion and low O_2 below, reaching extremes in mid-July 1972, diminished conditions suitable for zooplankton reproduction and survival within the water column.

Published information on effects of high pH on zooplankton is sparse. Davis and Ozburn (1969) report that the maximum pH at which *Daphnia pulex* survived was 10.4 in water of 70 mg HCO_3^- but reproduction occurred only up to pH 8.7. Limits in water of 10 mg HCO_3^- l^{-1} were narrower, 10.3 for survival and 8.2 for reproduction. O'Brien and DeNoylles (1972) report that 10.8 was acutely lethal to *Ceriodaphnia reticulata* in the laboratory. Mid-morning pH of 10.6 in ponds was associated with the disappearance of this species. Given the very soft water of Lake 227, it is likely that pH of 10.0 or above reflects conditions lethal for at least some of the crustaceans. Escape from high pH by remaining in the metalimnion would expose the zooplankton to low O_2. Some species of cyclopoids are reported to be able to withstand low O_2 or anoxic conditions (von Brand, 1944; Chaston, 1976). Cyclopoids appear to tolerate high pH better than cladocerans and calanoids judging from the pH limits given by Lowndes (1952). Resistance of cyclopoids to low O_2 and/or high pH may account for their better overall survival compared with cladocerans and calanoids.

An alternate hypothesis that predation by *Chaoborus* (Diptera:Chaoboridae) caused the decline of cladocerans and calanoids is discussed by Malley, et al. (In prep a.). Fertilization may have greatly enhanced the numbers of *Chaoborus* in Lake 227 but no quantitative data are available. Nevertheless, it is considered unlikely that predation would be the major cause of decline in these zooplankton species to such extremely low levels. Starvation and predation by fish are not considered to be important factors causing the decline. Edible algal species were abundant. Zooplankton was a minor food item of the fish in Lake 227 (fathead minnow, pearl dace, redbelly dace, finescale dace).

RESPONSE OF LAKE 223 TO ACIDIFICATION

Lake 223 is a small, oligotrophic lake with surface area of 27.3 hectares and maximum depth 14.4 meters Schindler, et al. 1980 and Schindler, 1980, describe the lake, the acidification scheme, and chemical and biological results from 1976 to 1979. Table 4 gives data on the pH and DIC content of the epilimnion of Lake 223 for 2 pre-acidification years and for the years 1976 to 1979 when sulfuric acid was added. DIC was greatly reduced in 1976, without lowering the pH. With the buffering capacity depleted, pH declined in response to acid addition in 1977 to 1979. Concentrations of P and N were not affected by acidification (Schindler, 1980). Phytoplankton biomass (Findlay and Saesura, 1980) and production (Schindler, 1980) increased during the first 4 years of acidification.

RESPONSE OF ZOOPLANKTON OF LAKE 223 TO ACIDIFICATION

The progressive acidification of Lake 223 from pH 6.8 in 1976 to an average of 5.60 during the ice-free season of 1979 has changed the abundance of certain zooplankton, particularly cladocerans and rotifers. Year-to-year variation in abundance of zooplankton species in the Experimental Lakes Area is not yet well-documented. Therefore, until trends reported here can be confirmed in 1980 or 1981, the conclusions are somewhat tentative. Seasonal abundances of species of crustaceans and rotifers in Lake 223 are described by Malley et al. (In prep. b.).

D. galeata mendotae declined during 1977 to 1979; on the other hand, *D. brachyurum* and *H. gibberum* were more abundant with acidification. *B. longirostris* remained relatively constant in numbers with acidification (Table 5).

The dominant calanoid, *Diaptomus minutus*, showed no change in numbers with acidification, whereas the minor species *D. sicilis*, disappeared in 1978. *Epischura lacustris* disappeared by 1979; *E. nevadensis*, not recorded in 1974, appeared in low numbers in 1977 and 1978 during early acidification but was not recorded in 1979. Overall, populations of calanoid nauplii and copepodids were relatively constant with acidification up to 1979 (Table 6). No effects of acidification on cyclopoids are evident (Table 7).

Total number of rotifers was higher in 1977, 1978, and 1979 than in the pre-acidification year, 1974. Most marked changes were increases in numbers of *Polyarthra vulgaris*, *P. remata*, *Keratella taurocephala*, and *Kellicottia longispina*.

Although species composition changed during 1977 to 1979, overall there has been no significant change in the biomass of crustacean zooplankton (Malley, unpubl. data).

Dramatic effects of acidification were observed on the population of the opposum shrimp, *Mysis relicta*, in Lake 223. This important fish food species in all but very deep lakes is found near the bottom by day and is planktonic at night, migrating vertically diurnally within the hypolimnion (Beeton, 1960). Population size in Lake 223 was estimated for the first time in summer 1978 as about 5.5×10^6 individuals. By summer 1979, the population was reduced to 10 percent of the 1978 numbers or less. In early 1978 the population tolerated a time when the maximum pH of their habitat was 5.9. In early 1979, the maximum pH which they found within their environment was 5.6. Thus the pH limit for survival of this species appears to be between 5.9 and 5.6 (Nero, unpubl. data)

Table 4. — Mean pH and DIC concentrations in the epilimnion of Lake 223 in the ice-free seasons of 1974 to 1979.

Year	Ice-free season epiliminion mean pH (range)	Range of DIC, epilimnion μ moles l^{-1}
Pre-acidification		
1974	6.64 (6.4-7.0)	100-150
1975	6.61 (6.5-7.0)	100-150
During acidification		
1976	6.79 (6.5-7.2)	40-100
1977	6.08 (5.6-6.3)	25-30
1978	5.84 (5.4-6.2)	20-30
1979	5.6 (5.4-5.8)	20-25

Table 5. — Number m^{-2} of cladocerans in Lake 223 averaged over the May to September period during 1974 and 1977 to 1979.

Species	1974	1977	1978	1979
Bosmina longirostris	10,850	10,800	12,200	4,450
Daphnia galeata mendotae	5,250	1,400	800	250
Holopedium gibberum	100	700	1,650	800
Diaphanosoma brachyurum	50	5,250	2,250	4,200

Table 6. — Number m^{-2} of calanoid copepods in L223 averaged over the May to September period during 1974 and 1977 to 1979.

Species	1974	1977	1978	1979
Diaptomus minutus adults	4,320	8,950	800	7,950
Diaptomus sicilis adults	650	100	0	0
Epischura lacustris adults	400	1,000	750	0
Epischura nevadensis adults	0	2,00	550	0
Calanoid nauplii N_1-N_6	54,050	67,100	46,100	48,600
Calanoid copepodids C_1-C_5	71,950	102,000	85,650	64,350

Table 7. — Number m^{-2} of cyclopoid copepods in L223 averaged over the May to September period during 1974 and 1977 to 1979.

Species	1974	1977	1978	1979
Cylops bicuspidatus thom. adults	3,650	11,500	7,900	8,700
Tropocyclops prasinus mex. adults	1,350	5,050	6,550	1,250
Mesocyclops edax adults	1,000	3,650	1,200	1,150
Cyclopoid nauplii N_1-N_6	51,850	102,000	155,750	58,350
Cyclopoid copepodids C_1-C_5	44,450	128,700	61,700	58,200

EFFECTS OF LOW pH ON BENTHIC AND PLANKTONIC CRUSTACEANS

A consistent result of surveys of zooplankton communities in lakes with a range of pH is that the number of species of crustaceans declines below pH 5.5 or 5.0 (Sprules, 1975; Leivestad, et al. 1976; Roff and Kwiatkowski, 1977). Daphnids are among the first to disappear (Almer, et al. 1974; Sprules, 1975). Raddum, et al. (1980) report fewer species of zooplankton in acidic lakes than in less acid lakes in Norway, with cladocerans suffering greater reduction than copepods and rotifers.

A number of benthic crustaceans, important as fish-food organisms, are sensitive to pH below 6.0. The amphipod *Gammarus lacustris* is absent from Norwegian lakes with pH below 6.0. The branchiopod *Lepidurus arcticus* is absent from these lakes below pH 6.1. In the laboratory, pH of 5.0 and below caused high mortality in adult *G. lacustris*. *L. arcticus* was affected at pH of 5.5 and below. Early life stages either did not survive or were delayed in molting (Borgstrom and Hendrey, 1976). Okland (1980) reports that *G. lacustris* tolerates pH down to 6.0 in colder mountain lakes but only down to 6.6 in warmer lowland lakes. The isopod *Asellus aquaticus* is rare below pH 5.6 in Norwegian lakes.

The amphipod *Gammarus pulex* in laboratory studies avoids water of below 6.2, or below 6.4 for young (Costa, 1967).

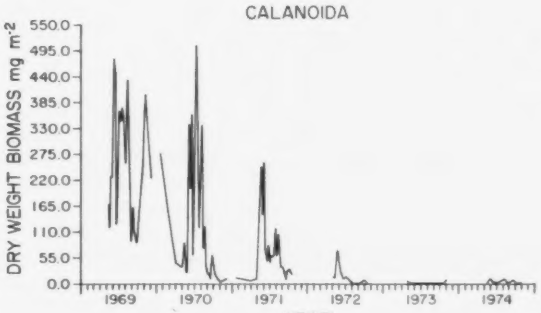
The sensitivity of these benthic crustaceans, and daphnids and *Mysis*, to the earlier stages of acidification, pH 5.5 and above, leads us to expect that acidification will have noticeable effects on fish species ecologically through the food supply as well as by direct physiological effects. The disappearance of these significant fish-food crustaceans from acidifying freshwater systems is thus an important early biological indicator of damage to the system.

The shifts in zooplankton species composition with acidification from large daphnids to smaller *Bosmina* and *Diaptomus minutus* leads Yan and Strus (In press) to conclude tentatively that filtering rates are lower in acidic than in non-acidic lakes. Raddum, et al. (1980) note that filtering zooplankton were proportionately more reduced in acidified lakes than were seizers. Acidification may thus affect the efficiency of energy transfer from primary to secondary trophic levels.

How low pH affects crustaceans is poorly known. Low pH may interfere with ion uptake (Sutcliffe and Carrick, 1973), particularly Ca^{++} uptake during postmolt (Borgstrom and Hendrey, 1976). Work on effects of low pH on Ca^{++} balance of postmolt crayfish *Orconectes virilis* was initiated at ELA to provide hypotheses concerning physiological effects of low pH on crustaceans which then could be tested on smaller planktonic and benthic species. All crustaceans molt periodically for growth and development and take up Ca^{++} from the environment after molting for recalcification of the new, soft exoskeleton (Greenaway, 1974, for crayfish; Marshall, et al. 1964, and Porcella, et al. 1967, for *Daphnia magna*). Postmolt *O. virilis* were found to be more susceptible to mortality at pH 3.0 and 4.0 than were non-molting individuals (Malley, 1980). Ca^{++} uptake during postmolt was progressively inhibited by pH below 5.75 and completely inhibited below pH 4.0 in crayfish in the laboratory in a known volume of lake water.

Although *O. virilis* in Lake 223 in 1980 at mean pH of 5.3 maintained pre-acidification population size and recruitment rate (Davis, unpubl. data), the exoskeletons of individuals from Lake 223 were not as hard in early fall 1980, as they were in control crayfish. Recalcification following molting in Lake 223 apparently is occurring at a rate slower than normal for these lakes. Rate of Ca^{++} uptake in postmolt *O. virilis* depends upon the concentration of HCO_3^- in the experimental medium, declining as the dissolved inorganic carbon equilibrium shifts away from HCO_3^- below pH 6.0 (Malley, 1980) and becoming greater as HCO_3^- is added to the medium of a crayfish or as the pH is experimentally raised to 8.0 or 9.0. (Malley, unpubl. data). Greenaway (1974) suggests that Ca is taken up partly in exchange for H^+ and partly accompanied by HCO^3 for electrical balance.

Crustaceans vary in their tolerance of low pH, some disappearing when environmental pH reaches only 6.0, others such as *Diaptomus minutus* and *Bosmina longirostris* existing in lakes at pH 3.8 (Sprules, 1975) or at 3.5 (*Mesocyclops edax, Cyclops bicuspidatus thom., Bosmina longirostris*, De Costa, 1975). Thus the pH range of 6.0 to 5.0 in which sensitive species of crustaceans decline in numbers and which noticeably affects the hardness of crayfish correlates with the decrease in HCO_3^-, as well as with an increase in H^+, both of which play a role in Ca^{++} uptake. It is interesting to speculate that in the acid-resistant crustacean species, ionic regulation, and particularly Ca^{++} uptake, depends upon exchanges with ions other than H^+ and HCO_3^-



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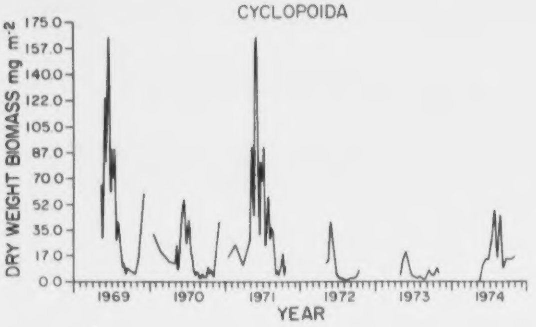

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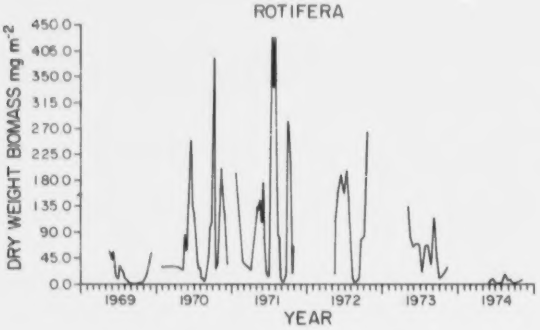

Figure 4. —

SUMMARY

Low alkalinity of these Experimental Lakes renders them prone to pH change from moderate levels of human activities such as nutrient addition or acid rain. Fertilization of Lake 227 at levels which increased P and N only five times above natural inputs, created adverse and unstable conditions for survival of zooplankton, including high pH and increased anoxia in the lake. The zooplankton community has not been able to recover pre-fertilization biomass or composition after 10 years of fertilization. Increased acidity to pH 5.6 has altered zooplankton species composition to a small extent but rate of loss of species is expected to be higher as the pH falls below 5.0 (Sprules, 1975; Roff and Kwiatowski, 1977). Until the acidification of Lake 223 progresses further, little can be said about the relationship between a species' ability to tolerate acid conditions and its ability to tolerate the high pH/lowO_2 conditions. *Daphnia* was sensitive to a reduction in pH to values below 6.0 and also declined in Lake 227 with elevated pH but the species were different in the two cases. Cyclopoids, tolerant of conditions in Lake 227 also were not affected by acidification in Lake 223.

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SEDIMENT TREATMENT FOR PHOSPHORUS INACTIVATION

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ABSTRACT

In situ sediment treatment has been studied to restore nutritionally polluted lakes. This method can alleviate many of the economic and environmental obstacles associated with dredging, bottom-sealing, or sediment consolidation via desiccation. A French experiment in progress since 1973 is based on improving the sediment sorptive capacity to adsorb phosphorus by injecting aluminum sulfate in its top layer. A prototype device designed for this purpose is described. The results indicate that the treatment significantly reduced the phosphorus in the lake even under anoxic conditions and during at least a 4-year period. No adverse long-term effects were observed. A Swedish experiment using nitrates, iron, and lime for sediment oxidation and phosphorus inactivation is also mentioned.

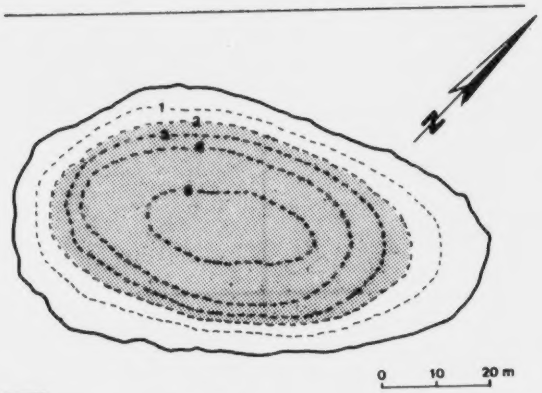
INTRODUCTION

As reported by Dunst and coworkers (1974) it is generally agreed that the most desirable approach to lake restoration is "to restrict the quantities of nutrients which reach the photic zone in a biologically available form at a time when they can contribute to the undesirable growth of aquatic plants." Curbing excessive nutrient inputs from the watershed is the most ecological and desirable long-term solution, but under certain circumstances the fertilizing power of the sediments is likely to delay or prevent lake restoration.

Different techniques are now available to reduce the sediment contribution to lake fertility (Theis, 1979). A new one is proposed which "defertilizes" sediments in the same way that agricultural techniques fertilize soils. Because phosphorus is a major eutrophicant and the easiest to render inactive (Vallentyne, 1974), a treatment was experimentally applied in summer 1973 to increase the phosphorus-binding capacity of the sediments, even under anoxic conditions (Barroin, 1976). This report summarizes the field experiment and its results.

EXPERIMENTAL LAKE DESCRIPTION

Lake Morillon is a doline lake located at 460 meters above sea level in the calcareous Chablais mountains bordering the south shore of Lake Geneva. Its main physical characteristics appear in Figure 1. Lateral inputs are diffused from the watershed which is mainly covered in gardens, lawns, and deciduous forest. Dead leaves that fall on the surface provide the lake with its mixotrophic characteristics such as yellow-brown water and loose organic sediments. There is no punctual outlet and the surface level is in equilibrium with the water table. Water and sediment chemistry, before treatment, is summarized in Tables 1 and 2. Phytoplankton was dominated by Dinophyta. The almost permanent presence of sulfides below a depth of 2 meters restricted the planktonic and benthic fauna to *Chaoborus flavicans* and the fish fauna to *Carassius auratus*.



▨ TREATED AREA : 1,800 m^2
TOTAL AREA : 3,500 m^2
MAXIMUM DEPTH : 5.5 m
VOLUME : 9,500 m^3

Figure 1. — Bathymetric map and physical characteristics of the experimental lake.

METHOD OF LAKE RESTORATION

Laboratory studies on sediments sampled at different water depths indicated that only those below 2 meters increased the fertility of epilimnetic water when mixed with it. The treatment was thus restricted to 1,800 m^2 and designed to affect the upper 15 centimeters of sediments, considered to be a thick enough barrier to

Table 1. — Water chemistry (before treatment).

		Surface		Bottom (20 cm above)	
Parameters	Units	Mini	Maxi	Mini	Maxi
Temperature	°C	0.0	25.5	4.2	11.0
Conductivity	μ S/cm (25°C)	300.0	500.0	680.0	550.0
pH	μ pH	7.4	8.9	6.4	6.9
O_2	mg O_2/l	0.5	20.0	$\leq$0.005	$\leq$0.05
S^{--}	mg S/l	$\leq$0.01	0.03	12.0	32.0
NO_3	mg N/l	$\leq$0.02	0.06	$\leq$0.02	$\leq$0.02
NO_2	mg N/l	<0.001	0.01	$\leq$0.001	$\leq$0.001
NH_4	mg N/l	0.02	2.55	3.9	33.0
PO_4-P	μg P/l	$\leq$1.0	104.0	30.0	2040.0
Tot-P	μg P/l	50.0	257.0	184.0	2380.0
Mg	mg Mg/l	4.2	5.5	5.6	7.2
Ca	mg Ca/l	53.0	98.0	105.0	150.0
Na	mg Na/l	3.6	4.2	3.9	4.6
K	mg K/l	3.7	4.5	3.4	4.6
Cl	mg Cl/l	3.7	6.0	4.2	14.0
SiO_2	mg SiO_2/l	2.4	10.3	13.0	32.0
SO_4	mg SO_4/l	16.8	27.8	0.0	26.8
Transparency	m	0.7	1.2		

Table 2. — Sediment chemistry (before treatment).

Parameters	% of dry weight - 105°
H_2O	900.0
Total P	0.2
Total K	0.2
Total Ca	20.0
Total Na	0.1
Total Mg	0.3
Total Mn	0.1
Total Si	2.0
Total Fe	0.7
Total Al	0.9
Org C	23.0
S^{--}	0.1
loss on ignition	
105° - 500° C	47
500 - 1000° C	20

prevent phosphorus migration from the lower layers. Aluminum sulfate ($Al_2(SO_4)_3\ 18H_2O$) was used because of the sorptive capacity of its hydroxide under anoxic and slightly acid conditions, its lack of acute toxicity, and the low price of the commercial product.

Laboratory studies revealed that an alum injection on the basis of 400 $g^{x}(m^2)^{21}$ would limit phosphorus release to an undetectable level. Figure 2 gives a schematic representation of the treatment equipment especially designed and constructed to inject alum into the sediment, minimizing perturbation of the water column stratification.

Epilimnetic water is pumped using a 12.5 HP pump, continuously receiving at the strainer level a 400 $g(m^2)^{-1}$ alum stock solution. A strong firehose conducts this diluted mixture to the ploughshare. This part of the equipment is made of a V-shaped iron tube fitted with regularly-spaced holes, the diameters of which are calculated so that the ejection pressure (c.a. $4kg(cm^2)^{-1}$) is the same for all. Pumps and reagent tanks are placed on a pontoon towed from the shore.

The treatment was applied during August 1973 in three phases: (1) The sediment was first ploughed, without alum, to degasify and prevent any subsequent lifting of flocculated materials by entrapped gas bubbles; then (2) 750 kilograms of alum were injected in the sediments; and (3) finally, 200 kilograms poured out on the whole surface for precipitation sediment particles previously mobilized, thus performing an epilimnetic inactivation.

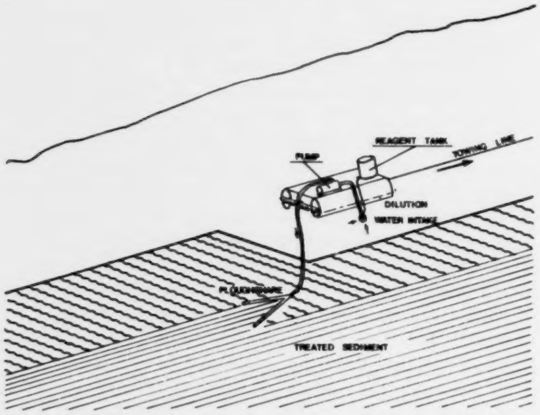


Figure 2. — Schematic representation of the treatment equipment.

RESULTS

Transparency

For a few days after treatment, Secchi disk readings increased by up to 3 meters; later they decreased again because of the rising and dispersion of a few sediment flocs. Final result: There has been a slight increase of the mean value and a greater amplitude of the variations.

WATER CHEMISTRY

PO_4-P and Tot-P concentrations drastically decreased at the sediment interface (Figure 3) as well as

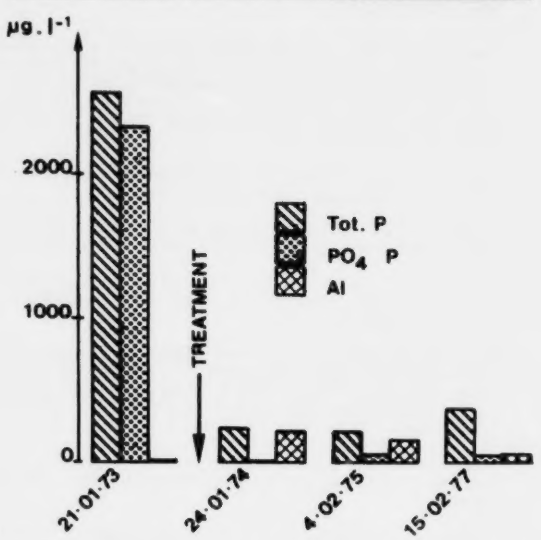


Figure 3. — Concentrations of Tot-P, PO_4-P and A1 in water at sediment interface.

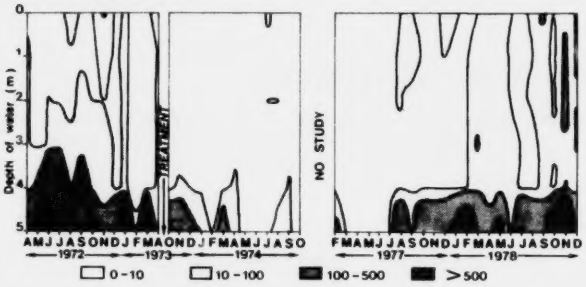


Figure 4. — Distribution of PO_4-P ($\mu.l^{-1}$) over time and depth.

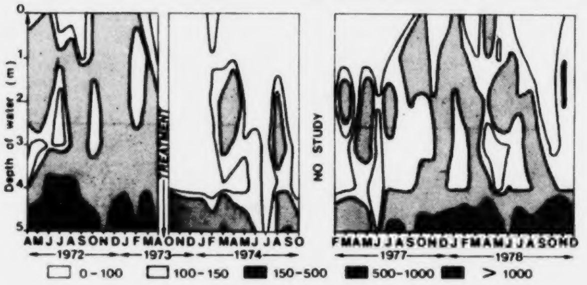


Figure 5. — Distribution of Tot-P ($\mu.l^{-1}$) over time and depth.

in the whole lake (Figures 4,5) until 1977. Simultaneously, aluminum concentrations increased to detectable levels; likewise, sulfides' concentrations in bottom layers rose to 110 $mgSl^{-1}$ in 1974, falling to 57 57 $mgSl^{-1}$ in 1977. No other significant change was noticed concerning water chemistry.

WATER BIOLOGY

During 1974, phytoplankton biovolume showed a 50 percent reduction accompanied with a specific shift from Dinophyta and Chlorophyta to Cryptophyta and Diatoms (Figure 6). During 1977-1978 Chrysophyta and Diatoms were dominant. No adverse effects were observed concerning the originally scarce fish and plankton fauna.

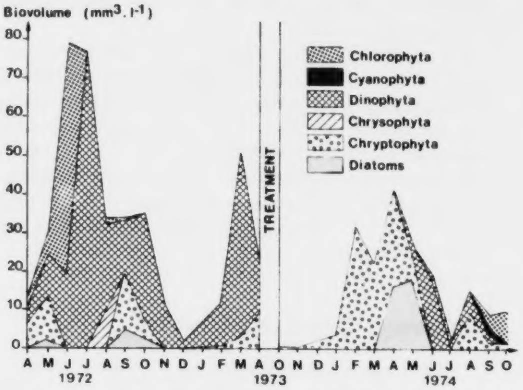


Figure 6. — Quantitative and qualitative evolution the phytoplankton.

SEDIMENT CHEMISTRY

For a few days after treatment, many aluminum hydroxide flocs were observed at the sediment interface probably resulting from the epilimnetic inactivation. By studying cores sampled a few weeks after treatment, no visual evidence of sediment mixing could be detected and no significant change in the distribution patterns of studied elements, among them aluminum, could be measured.

MACROPHYTES

During the year after treatment, some nenuphars showed irregular foliation but in 1975 no anomaly was observed.

DISCUSSION

To properly evaluate the importance of treatment efficiency and duration it is necessary first to evaluate that of the part played by epilimnetic inactivation. In fact, the introduction of 200 kilograms of alum in a 9,500 m³ volume, produces a final aluminum concentration of 1.9 mgl^{-1}, which represents about a tenth of the amount indicated in the literature (Funk and Gibbons, 1979). Therefore, it may be assumed that the success in lowering phosphorus content of the whole lake during several years are due chiefly to sediment treatment. The phosphorus-binding capacity of the treated sediment seems to show a long-term saturation as indicated from phosphorus concentration increases during 1978, perhaps because of constant input including, for example, untreated littoral sediments or dead leaves.

Owing to the important buffer capacity of the water and the sediment, no pH modification was observed after alum had been injected, the reaction of which is acid. The slight increase of the dissolved aluminum concentration indicates that this element was totally in a flocculated form. It is surprising that the analysis of the cores did not show any significant increase. Several explanations may be suggested: the reagent was injected in a higher thickness than planned and therefore was more diluted, or the sediment aluminum content was high and variable enough to preclude any observation of change or to make the sampling inadequate.

The sulfates introduced in the sediments were reduced to sulfides by the sulfato-reducing bacteria and then migrated mostly to the hypolimnion increasing its sulfide content. The slight oligotrophication indicated by phytoplankton changes cannot be only interpreted as resulting from the treatment because of the possible interference of additional environmental factors; but the nenuphar disease is due to rhizome deteriorations and perhaps nutrient deficiency directly connected with manipulation of the lake bottom.

FURTHER DEVELOPMENTS

The Morillon experiment having been relatively successful, further research was done to construct more elaborate equipment, as autonomous as possible. An intermediary prototype will be tested during summer 1980. Other research is being conducted to investigate the efficiency of different chemicals for increasing the phosphorus-binding capacity of sediments directly, or through oxidization using peroxides.

Meanwhile, in 1975, a Swedish experiment was conducted on a larger scale, using iron for phosphorus fixation and nitrates for sediment oxidization after pH adjustment with lime (Ripl, 1976; Bjork, 1978; Bjork, et al. 1978). The harrow especially designed for this purpose lifted the sediment using compressed air, chemicals being simultaneously distributed at the rear of the device. After addition of nitrates, denitrification processes took place, producing a vigorous release of nitrogen bubbles. As Dr. Bjork said, "thanks to the treatment Lake Lillesjon was converted to a lake with normal ecosystem functions."

CONCLUSION

Sediment treatment methods open up an important field for applications which make it possible to control sediment conditions and therefore the state of the entire lake. This control concerns not only phosphorus fixation with or without oxidization but also every phenomenon occurring in the sediments. The same Swedish researchers, in collaboration with the private firm Atlas-Copco, are now developing the "Contracid method" to counteract lake acidification (Bjork, 1978). The harrow used for injecting an alkaline sodium solution has a capacity of about 10 times that of the prototype used for sediment manipulation in Lake Lillesjon.

It is easy to imagine using such techniques for solving problems of oil pollution by injecting cultures of "trained" bacteria, or of macrophyte proliferation by injecting selected biocides. But sediment treatment is only at its neolithic stage and requires not only more technological research but also more limnological knowledge.

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TWO EXAMPLES OF URBAN STORMWATER IMPOUNDMENT FOR AESTHETICS AND FOR PROTECTION OF RECEIVING WATERS

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ABSTRACT

Stormwater impoundments in urban areas can improve the quality of runoff prior to discharge to the receiving waters, and at the same time, aesthetically improve the urban environment. Two manmade lakes of contrasting design that are fed only by stormwater runoff are examined. Lake Aquitaine has a small drainage basin and a sophisticated sediment removal system; Lake Wabukayne has a large catchment area and only a gabion wall for sediment removal. The long-term average retention time of the Lake Aquitaine sedimentation basin, 17 days, is similar to the retention time of the whole of Lake Wabukayne, 13 days. Suspended solids are reduced by 69 to 90 percent across Lake Aquitaine, while the solids reduction across Lake Wabukayne (29 to 33 percent) is similar to the reduction across the sedimentation basin in Lake Aquitaine (4 to 25 percent). *Cladophora* has become the dominant alga in Lake Aquitaine while in Lake Wabukayne aquatic macrophytes, *Cladophora*, phytoplankton and floating algae are prevalent. Both lakes have reduced dissolved oxygen levels at all depths following heavy rainfalls. Lake Aquitaine was anoxic at 1 m above the bottom in August 1980. The retention time of the lakes and sedimentation basin appears to be the main factor controlling the reduction of suspended solids. Eutrophication problems may require further control measures to maintain the aesthetic value of the lakes.

INTRODUCTION

The Province of Ontario has an estimated 500,000 lakes providing a huge potential for water based recreation. The high quality lakes on the Precambrian Shield support a large tourist industry and have been the recreational playground for many residents living in urban areas such as Toronto and Hamilton. The recreational value and water quality of the smaller number of lakes closer to the urban areas in Southern Ontario have not been studied as much. However, in recent years an increased awareness and concern for the quality of the urban environment has produced demand for high quality lakes close to and within the urban areas. This demand is reflected in two ways: A need to protect and upgrade water quality in existing urban waterways and a desire to create new lakes within city limits as aesthetic improvements to the urban environment.

Urbanization itself causes deterioration of receiving water quality by increasing the total amount and peak loading of runoff as well as adding a wide range of contaminants to the runoff (Weibel, 1969).

Using impoundments to improve the quality of the runoff before it enters the receiving waters is an attractive concept; it is logical to design such impoundments to meet some aesthetic demands as well. This type of lake is inexpensive to maintain compared to an equivalent amount of parkland (Proj. Plan. Associates, 1976) and does not constitute a safety hazard provided it is well designed and properly maintained, with certain use restrictions.

This paper discusses two impoundment lakes of contrasting design.

Study Lakes

Lake Aquitaine is located in the Meadowvale "new town" in Mississauga about 20 kilometers west of Toronto. Construction began in the 1970's and this agricultural land is still being developed for housing, shopping, and light industry. The town covers 1,200 hectares and is expected to have a population of 65,000 when complete in 1985. Its proximity to the Credit River and the impoundment lake have been strong selling features used by the developers.

Lake Aquitaine was constructed by excavating a farm field to a depth of 5 meters with a bank slope of 4:1 at the shore. A concrete sedimentation basin, with energy dissipation system, surface skimming weir, and perforated spillway, were constructed at the inlet. A "morning glory' spillway including bottom draw facilities, controls water levels in the lake and an emergency spillway and drainage channel have been provided as a precaution in the event of a hurricane-like storm. An aerial view is shown in Figure 1 and lake characteristics are shown in Figure 3 and Table 1. Discharge is to the Credit River system. The main inlet is a single 2 m × 3 m storm drain. There is a supplementary water supply from a 10 centimeter (4-inch) municipal water main to maintain water level in

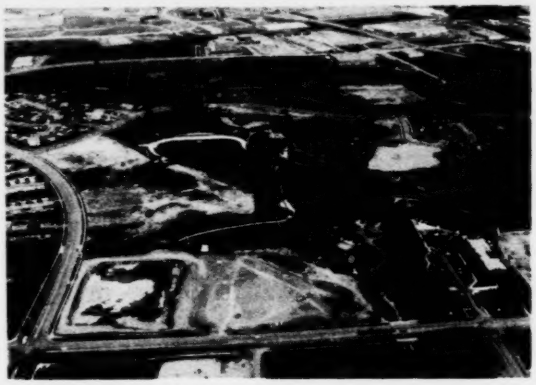

Figure 1. — Aerial photograph of Lake Aquitaine (eastward view).

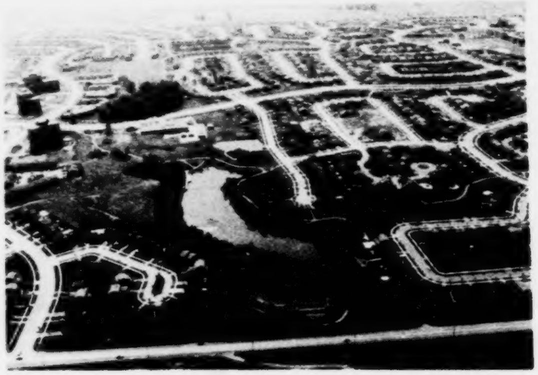

Figure 2. — Aerial photograph of Lake Wabukayne (westward view).

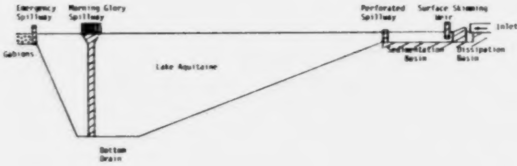

Figure 3. — Diagrammatic representation of longitudinal section through Lake Aquitaine (not to scale).

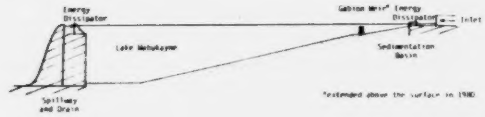

Figure 4. — Diagrammatic representation of longitudinal section through Lake Wabukayne (not to scale).

the summer if necessary. Since the lake was entirely manmade, it was designed to maximize both water quality improvement and aesthetic value. Model predictions were made on the sedimentation basin and lake characteristics (Murrey and Ganczarczk, 1977). Construction was completed during the winter of 1976-1977 and the maximum water level was reached in mid-August 1977. At the same time sodding and tree planting were completed on the surrounding parkland. The area immediately around the park is currently under development.

Regulations prohibit activities requiring primary contact (e.g. swimming, bathing, wind-surfing etc.) and power boats are not permitted. However, canoeing, sailing, and paddleboating are encouraged.

Lake Aquitaine was stocked with 3,300 rainbow trout *Salmo gairdneri* (10 to 30 cm) in September 1977 and again in September 1979 with ≈500 rainbow trout. Huge schools of small minnows have been observed near the marina, and a large population of pumpkinseeds *Lepomis gibbosus* has become established in the lake. Fishing is permissible subject to Ministry of Natural Resources fishing regulations for Peel Region. In September 1979 a fishing derby was organized at Lake Aquitaine.

Lake Wabukayne is located in the Erin Mills development of Mississauga about 1 km from Lake Aquitaine. Prior to development, a farm pond had been built in the steep-sided valley of Wabukayne Creek. The topography greatly limited possible design changes for the lake. The farm pond was drained and lined with clay. An aerial view is shown in Figure 2 and the lake characteristics are shown in Figure 4 and Table 1.

Table 1. — Characteristics of the impoundment lakes.

	Aquitaine	Wabukayne
Lake area (ha)	4.7	2.0
Lake volume (m^3)	1.8×10^5	3.2×10^4
Mean depth (m)	3.8	1.6
Maximum depth (m)	5.0	2.9
Sedimentation basin area (ha)	0.38	0.27
Long term average yearly flow (m^3)[1]	2.0×10^5	9.3×10^5
Retention time of the lake (days)	329.0	13.0
Retention time of the sedimentation basin (days)	17.0	0.55
Drainage basin area (ha)		466.0
developed (ha)	48.0	263.0
undeveloped (ha)	59.0[2]	203.0[3]

[1] Average for 10 years period for fully developed watershed (3)
[2] 34 ha being developed in 1980
[3] 184 ha being developed in 1980

A large concrete dam and spillway replaced the original earth dam and the lake was re-filled by the fall of 1976. The lake receives inputs from several storm sewers and Wabukayne Creek; all are channeled through twin 3.0 m diameter storm sewers and a single 1.6 m sewer. A 6.3 l/sec supplementary supply of water is pumped to the lake from groundwater sources to help maintain lake levels. Discharge is to the Credit River. The sedimentation basin in Lake Wabukayne was separated from the main lake by a submerged gabion weir near the inlet end of the lake and an access ramp for sediment removal was constructed. In 1980, another row of gabions was added to bring the weir above the waterline. A 3 m gap was left at each end of the weir to discourage children from playing on it. A small park with some naturally wooded areas borders the shoreline; most of the land is sodded, but erosion does occur in the steep-banked areas (slope 2:1). The area adjacent to the park is completely developed, but the remainder of the watershed is a mixture of developed, developing, and rural land. No primary contact recreation or boating is permitted. Fish stocking has not been undertaken but natural populations of sticklebacks (*Gasterosteidae*) have been observed. Tadpoles, frogs, crayfish, and leeches also seem to thrive.

Methods

Samples are collected weekly from May to September and less frequently in October.

Sampling at mid-lake stations included Secchi disk depths, temperature profiles at the deepest point in each lake, and dissolved oxygen at two depths. Chemical samples were collected as composites through the euphotic zone or to 1 m above bottom using a plastic hose or a weighted sampling can. A separate sample was collected at 1 m above bottom if the euphotic zone did not penetrate to this depth.

Water samples were collected at the inlets and outlets of the lake and sedimentation basin of Aquitaine whenever flow was sufficient and at the inlet and outlet of Wabukayne. It was not possible to sample at the outlet of the Wabukayne sedimentation basin since it was defined only by a submerged barrier.

Fish samples for heavy metal analysis were collected from both lakes by the Ontario Ministry of Natural Resources in September 1979.

All analyses were performed at the Ontario Ministry of the Environment Laboratory according to their standard methods (Outlines of Analytical Methods, 1975).

RESULTS AND DISCUSSION

The watersheds are not yet fully developed and the construction activity will continue to influence the quality of the runoff so the data represent a transition stage for the lakes. However, a number of observations can be made regarding the ability of the lakes to achieve their two main objectives: Aesthetics and receiving water protection.

The main contrasting feature of the lakes is the retention times. The long-term average retention time of the Aquitaine sedimentation basin, 17 days, is similar to the retention time of the entire Lake Wabukayne, 13 days (Table 2).

Mineral Chemistry

The major ion content of both lakes is shown in Table 2 for early June 1979.

With the exception of sodium and chloride, the major ions are similar, reflecting similar soil conditions. These values are typical of the sampling period for all 3 years.

Sodium chloride from road de-icing dominates the ion content of Lake Aquitaine. Thousands of tons of salt are used each year by the city of Mississauga and these watersheds no doubt receive a share along with contributions from householders treating driveways and sidewalks. It is not clear why salt concentrations in Lake Wabukayne are much lower since the same percentage of the watershed is developed.

Table 2. — Major ion content of Lakes Aquitaine and Wabukayne on June 14, 1979. Units are in meq/l except conductivity (in μmhos/cm at 25^{0}C) and pH.

	Aquitaine	Wabukayne
Ca^+	2.1	2.1
Mg^+	0.67	1.0
Na^+	5.4	1.6
K^+	0.11	0.17
Cl^-	6.2	1.8
$SO_4^=$	0.89	1.0
Alkalinity (HCO_3^-	1.7	2.3
Cond.	1000.0	530.0
pH	8.24	8.06

The chloride concentrations and conductivity decrease during the summer in both lakes although salt content has generally increased, particularly in Aquitaine, Figure 5.

Suspended Solids and Turbidity

One of the prime functions of the impoundments is to reduce the solids loading to the receiving water. Table 3 shows the performance of both lakes in this respect.

In Lake Aquitaine, very good reduction of solids is occurring across the whole lake; the sedimentation basin itself is not so effective. However, about 20 centimeters of black sludge has accumulated in the basin since it was built.

Lake Wabukayne is achieving an overall reduction in solids similar to the Aquitaine sedimentation basin; this is not surprising since they have similar average retention times.

The model projected a fivefold increase in suspended solids discharge from the Aquitaine watershed following development (Murrey and Ganczarczyk, 1977). It further projected that the lake and sedimentation basin combined would give a 93 percent reduction. While it is not possible to draw a comparison between the projected loadings and the observed concentrations, it would appear that Lake Aquitaine is very close to meeting its suspended solids objective since the reductions in solids concentrations ranged from 69 to 90 percent.

Results for August 24 and 31 were left out of the 1978 data set averaged for Table 3. A prolonged dry spell followed by some rain produced high suspended solids of 261 and 1,367 mg/l, respectively, in the low inlet flows. These values are one to two orders of magnitude greater than recorded for any other sampling date. Including them in the yearly average distorts the impression of the effectiveness of solids removal.

Turbidity data generally paralleled the suspended solids results. The data for 1979 are shown in Figure 6 along with the total rainfall since the previous sampling date. Lake Wabukayne is always more turbid than Aquitaine with a greater response to rainfall. The

Table 3. — Effects of the impoundments on water quality (in mg/l).

	Year	Inlet	Overflow from sedimentation basin	Percent reduction	Lake	Outlet	% reduction from the inlet
Lake Aquitaine							
Suspended Solids	1977	57.0	48.0	15.0	10.0	5.8*	90
	1978	11.0	8.0	25.0	3.8	3.4	69
	1979	10.0	9.8	4.0	2.0	1.5	85
Nitrogen	1977	2.41	2.31	-	1.84	1.84	25
	1978	2.37	2.33	-	0.93	0.98	58
	1979	1.79	1.98	-	1.01	1.05	39
Total phosphorus	1977	0.20	0.20	-	0.039	0.045*	78
	1978	0.17	0.16	-	0.036	0.032	81
	1979	0.10	0.10	-	0.023	0.022	79
Lake Wabukayne							
Suspended Solids	1977	16.0	-	-	14.0	12	29
	1978**	13.0	-	-	13.0	8.5	33
	1979	15.0	-	-	11.0	11	29
Nitrogen	1977	2.23	-	-	2.21	2.24	0
	1978**	1.42	-	-	1.12	1.21	15
	1979	1.90	-	-	1.53	1.43	25
Total Phosphorus	1977	0.060	-	-	0.071	0.070	-
	1978**	0.050	-	-	0.037	0.044	12
	1979	0.89	-	-	0.068	0.065	27

* only 3 data sets
** 2 data sets left out, see text

results reflect the retention time of the lakes. On a number of sampling dates, the turbidity at the Wabukayne outlet was nearly the same as at the inlet from the storm sewers while turbidity at the Aquitaine outlet was consistently lower than at the inlet. For example, on June 14, 1979 the inlet and outlet values for Aquitaine were 16 and 2 Formazin Turbidity Units (F.T.U.), respectively, and for Wabukayne they were 84 and 86 F.T.U. respectively.

Lake Wabukayne would have to be increased in volume by 25 times to give a retention time equal to Aquitaine; this may be necessary to achieve an effective suspended solids control. However, such a large increase in size would not be practical in this particular case.

Nutrients and Eutrophication

Although nutrient levels in both lakes have been high enough to produce nuisance growths of algae, Lake Aquitaine has never had a serious phytoplankton bloom, and Lake Wabukayne only began to exhibit high chlorophyll *a* concentrations in 1979 (Tables 3 and 4). Chlorophyll concentrations were not reported for 1977 because of an analytical problem.

In Lake Aquitaine the macrophyte, *Potamogeton foliosus*, was first noticed in August 1978. At the same time small colonies of attached algae (*Oedogonium* and *Cladophora*) began to develop. By 1979 *Cladophora* occupied almost all available substrate; only the finer gravel remained devoid of the alga. A band of *Cladophora* currently extends around the entire shoreline of the lake. Raking has become necessary to remove the alga to prevent odors from decomposition. Other algal types identified near the inlet end include *Closterium*, *Spirogyra*, *Synedra*, and *Rhizoclonium*. Attached algae have flourished, and by successfully competing for available nutrients, may have caused a decrease in free-floating algae in 1979, as measured by chlorophyll *a*.

In Lake Wabukayne a wide variety of plant life was established as early as June 1977. *Polygonum* sp., *Typha* sp., and *P. zosteriformis* began to develop in an area of the north shore just below the sedimentation basin. By July 1977 *Alisma plantago-aquatica* occupied most of the shoreline including the sedimentation basin. The following summer dense mats of floating algae (*Spirogyra* and *Oscillatoria*) began to develop over the macrophyte beds. Oil and floating debris tended to collect in these mats, further detracting from the appearance of the lake. *Cladophora* growths were observed on the gabions and on all concrete surfaces near the outlet. Conditions continued to deteriorate when a relatively wet spring and early summer in 1979 resulted in highly turbid conditions and poor water clarity. Macrophyte growth was thus restricted, but phytoplankton levels began to increase, producing chlorophyll *a* levels as high as 50 μg/l (July 10). By late July drier weather prevailed and water clarity improved as turbidity and suspended solids levels decreased. At this time many large *Daphnia* sp. were observed which could have contributed to the decrease in Chlorophyll *a* that occurred (2.2 μg/l, August 8). As in 1978, the improved light conditions promoted the growth of algal mats over the macrophyte beds. *Cladophora* continued to grow where suitable substrate was available but lacked the "healthy" appearance of the *Cladophora* in Lake Aquitaine.

The combination of solids removal and plant growth is substantially reducing phosphorus in Lake Aquitaine but having a minimal effect in Lake Wabukayne, Table 3.

The steadily deteriorating conditions in Lake Wabukayne prompted a public meeting in the fall of 1979 which resulted in the clearing of the sedimentation basin and the modifications to the gabion wall. A regular program of surveillance and debris removal was also initiated.

Dissolved Oxygen

Dissolved oxygen levels in the surface waters of both lakes have generally been adequate although there have been brief periods of reduced oxygen conditions in the bottom waters of both lakes. In Lake Aquitaine the bottom water dissolved oxygen levels have progressively deteriorated from 1977 through 1979. By August of 1980 anoxic conditions were measured at 1 m above bottom and hydrogen sulfide was present.

Both lakes suffer from short-lived reduced oxygen conditions at all depths following heavy rainfall. The effect is most pronounced in Lake Wabukayne where higher sediment loads produce higher oxygen demands. Lowest observed oxygen concentrations in the surface waters have been 5.4 mg/l and 3.7 mg/l in Aquitaine and Wabukayne, respectively.

Fishery

Nine samples of rainbow trout from Lake Aquitaine and seven samples of sticklebacks from Lake Wabukayne were analyzed for PCB's, Mirex, pesticides, mercury, copper, nickel, zinc, lead, cadmium, chromium, arsenic, selenium, and iron. The rainbow trout were in the 30 to 46 cm (12 to 18 in.) size range and in all cases were acceptable for unrestricted consumption. The Lake Wabukayne sticklebacks were all less than 15 cm (6 in.) and, although unlikely to be consumed by humans, were similarly low in contaminants.

Angling for rainbow trout has been a popular event in Lake Aquitaine since the first stocking of fish. The fishing derby in 1979 was a great success with numerous fishermen taking part, again emphasizing the recreational potential and aesthetic value of the lake.

Conclusions

Properly designed stormwater impoundments can effectively protect the quality of receiving waters and provide aesthetic value to the urban environment.

Good control of suspended solids has been achieved at an average retention time of 329 days while a 13-day average retention time gives a very limited control.

Problems related to eutrophication of the impoundments seem to be increasing with time and may require control measures in the future; otherwise, the aesthetic value of the lakes may be reduced.

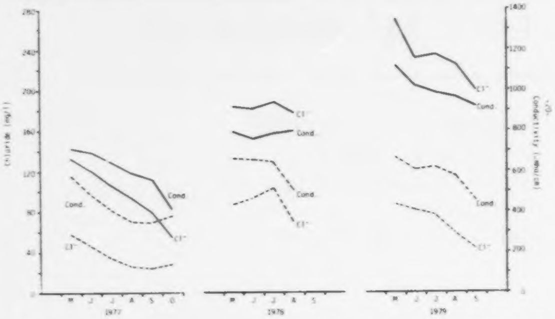

Figure 5. — Summary of monthly mean chloride and conductivity levels for 3 years (1977-79). Solid line represents Lake Aquitaine; broken line represents Lake Wabukayne.

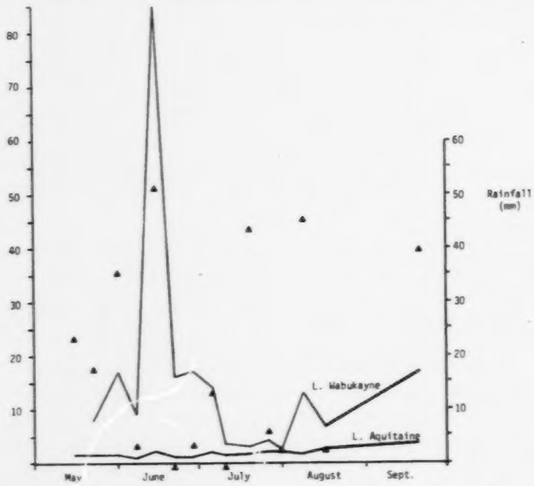

Figure 6. — Summary of turbidity and precipitation levels from May to September, 1979. Black triangles indicate the total amount of rainfall in the previous week.

Table 4. — Chlorophyll *a* concentrations in Lakes Aquitaine and Wabukayne in 1978 and 1979 in μg/l.

		Mean	Range
Lake Aquitaine	1978	6.0	1.0-16.6
	1979	3.4	1.0- 8.2
Lake Wabukayne	1978	6.8	0.7-34.4
	1979	23.0	1.8-50.0

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REVIEW OF AERATION/CIRCULATION FOR LAKE MANAGEMENT

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ABSTRACT

Artificial circulation is a management technique for oxygenating eutrophic lakes subject to water quality problems, algal blooms, and fish kills. Whole lake mixing may reduce regeneration of nutrients from profundal sediments, while often controlling blue-green algal blooms. Models predict that overall algal biomass will decrease in deeper lakes when light limitation is induced by mixing. If destratification elevates epilimnetic CO_2 levels and causes a sufficient drop in pH, dominance in the algal community will likely shift from a nuisance blue-green species to a mixed assemblage of green algae. This more edible resource combined with an expansion of habitat leads to more abundant zooplankton and with provisioning of a hypolimnetic refuge, invasion of large-bodied daphnids. Habitat expansion and shifts in community structure of benthic macroinvertebrates potentially elevates the abundance of fish food organisms. Although short-term increases in fish growth and yield have been attributed to improvements of food and habitat resources, documentation of long-term changes is lacking. In southern regions, artificial circulation provides benefits for warmwater fishes only.

INTRODUCTION

Artificial aeration or circulation of lakes is commonly used for managing the ecological consequences of eutrophication. By inducing dramatic changes in species abundance and distribution, diversity, and trophic structure, the technique has potential usefulness in controlling algal blooms and improving fisheries. This paper examines artificial circulation techniques; i.e., those that mix the whole lake and provide aeration without attempting to preserve the normal thermal structure. Hypolimnetic aeration, which maintains aerobic conditions without disrupting thermal stratification, is covered elsewhere (Fast and Lorenzen 1976; Pastorok, et al. in press).

EFFECTS OF ARTIFICIAL CIRCULATION ON WATER QUALITY

Chemical Parameters

In most cases, artificial destratification increases the concentration of dissolved oxygen in bottom waters immediately (e.g., Hooper, et al. 1953; Lackey, 1972; Haynes, 1973). Dissolved oxygen in the former epilimnion may show a corresponding decrease due to reduced photosynthesis (Haynes, 1973) or mixing of hypolimnetic waters with low dissolved oxygen and high BOD into the surface layer (Ridley, et al. 1966; Thomas, 1966). Over a period of several weeks, the oxygen content of the whole lake increases (Pastorok, et al. in press). Under some circumstances, oxygen depletion cannot be prevented by normal levels of artificial aeration and massive fish kills result (R. S. Kerr Res. Center, 1970; McNall, 1971).

Oxygen levels influence redox reactions involving Fe, Mn, and Al; in turn, these elements and their complexes partly determine the availability of nitrogen and phosphorus compounds through release processes occurring at the surface of profundal sediments (Mortimer, 1941, 1942; Holdren and Armstrong, 1980).

As hypolimnetic waters are brought to the lake's surface, excess gases such as CO_2, H_2S, and NH_3 are released to the atmosphere (R. S. Kerr Res. Center, 1970; Toetz, et al. 1972; Haynes, 1973). Along with oxygen and other chemical species, these gases become isochemical with depth (Toetz, et al. 1972).

Transparency

Artificial circulation has varied effects on water transparency, depending on the intensity of mixing and the contribution of phytoplankton to turbidity levels before treatment. When mixing is induced during a surface bloom of blue-green algae, transparency will increase immediately due to distribution of the algae throughout a greater water volume (Haynes, 1973). Thereafter, water clarity may be enhanced by destruction of the bloom through light limitation in deep lakes (Lorenzen and Mitchell, 1975) or through a change in some other environmental factor in shallow lakes (Malueg, et al. 1971).

A decrease in transparency after mixing generally correlates with a rise in total seston (Garton, 1978; Garton, et al. 1978), which may be caused by surface algal blooms (Hooper, et al. 1953; Drury, et al. 1975) or resuspension of sediments (Fast, 1971a). Most destratification devices have been undersized with respect to the scaling rule suggested by Lorenzen and

Fast (1977, i.e., 9.2 m^3/min per 10^6 m^2 lake surface. When more thermal energy is absorbed at the lake's surface than the circulation device can distribute, then microthermal stratification of 2 to 3°C provides algal populations a surface refuge with high light levels (e.g., Fast, 1973a; Drury, et al. 1975).

EFFECTS OF ARTIFICIAL CIRCULATION ON PHYTOPLANKTON

In 40 cases of complete destratification, only 65 percent (=26 experiments) led to any significant change in algal concentrations; of these, about 30 percent resulted in more algae. Table 1 summarizes the responses of phytoplankton to artificial circulation for each lake. When more than one experiment was conducted in a lake, the predominant response is given unless the data are too variable to indicate an overall trend; then, the responses for individual experiments are given. Where mixing was complete, aeration decreased algal density or biomass in 13 of 23 lakes. In three lakes, the amount of phytoplankton remained about the same, and in seven lakes it increased or the overall response was unclear. Where mixing was incomplete, algal density generally stayed the same or

Table 1. — Responses of phytoplankton to artificial circulation[a]

Lake	Reference	Algal Density[c]	Algal Standing Biomass[d]	Mean Chlorophyll-*a* Concentration	Green Algae	Blue-green Algae	Ratio Gr:Bl-gr
Complete Mixing							
Cline's Pond	Malueg, et al. 1971	-		-	0	-	+
Parvin Lake	Lackey, 1973a	-			-	0[e]	0
Section 4 Lake	Fast, 1971a	-f	-		-		
	Fast, et al. 1973						
Boltz Lake	Symons, et al. 1967, 1970	-	-		-	-	+
	Robinson, et al. 1969						
University Lake	Weiss and Breedlove, 1973	-		0	+	-	+
	Haynes, 1973	-				-	+
Kezar Lake	N.H.W.S.P.C.C.1971		-	0	+		
	Lorenzen and Mitchell, 1975						
King George VI	Ridley, et al. 1966	-			-	+	-
Indian Brook[b]	Riddick, 1957	-					
Prompton Lake[b]	McCullough, 1974	-					
Cox Hollow[b]	Wirth and Dunst, 1967	-					
	Wirth et al. 1970						
Stewart Lake	Barnes and Griswold, 1975			-			
U.K. Reservoir[b]	Ridley, 1970					-	
Wahnbach Reservoir	Bernhardt, 1967					-	
Queen Elizabeth II		0					
		-					
Lake Roberts	McNall, 1971	+				+	
	R.S. Kerr Res. Cen., 1970	-				-	
Falmouth Lake	Symons, et al. 1967, 1970	+				-	+
	Robinson, et al. 1969	+			+	-	+
Test Res. II	Knoppert, et al. 1970	+	+		0	+	0
Buchanan Lake	Brown, et al. 1971	+	+	+	+	-	+
Ham's Lake[f]	Steichen, et al. 1974			0	0	0	0
	Toetz, 1977a, b	0					
	Garton, 1978						
Test Res. I	Knoppert, et al. 1970	0+	0+			0+	0-
Mirror Lake	Smith, et al. 1975	0[g]	0[g]			0[g]	
4 Lakes[h]	Irwin, et al. 1966	0				-	+?
Starodworskie Lake[f]	Lossow, et al. 1975				-	+	-
Incomplete Mixing							
Casitas Res.[b]	Barnett, 1975	-				-	
Hyrum Res.	Drury, et al. 1975	+	+	+	-	+	-
West Lost Lake	Hooper, et al. 1953	+	+			+	
Pfaffikersee	Thomas, 1966	+				+	
Waco Res.[b]	Biederman and Fulton, 1971	0			0		
Lake Maarsseveen[b]	Knoppert, et al. 1970	0			0		
Lake Catharine	Kothandaraman, et al. 1979	0					0
El Capitan[f]	Fast, 1973a	+?					0
Arbuckle Lake	Toetz, 1977a, 1979	0		0			0
Lake Calhoun	Shapiro and Pfannkuch, 1973		+	+	0	+	-

[a] + = decrease, 0 = no significant change
[b] qualitative information only
[c] cells or colonies per liter; weighted mean for water column unless noted
[d] weight per square meter of lake surface
[e] increase observed, but control year was unusual
[f] samples were taken near lake surface
[g] increase observed, but it was correlated with large input of allochthonous nutrients
[h] Stewart Hollow Lake, Caldwell Lake, Pine Lake, Vesuvius Lake

increased following treatment (Table 1). Although artificial circulation usually has a negative influence on blue-green algae, its effect on green algae is ambiguous.

Physical Mechanisms

In lakes where algal production is potentially limited by light, several models predict a decrease in net photosynthesis and a reduction in standing crop of algae as depth of the mixed layer increases (e.g., Lorenzen and Mitchell, 1975; Oskam, 1978). If algae are limited by nutrients before circulation, however, a slight increase in mixing depth could cause an elevation of standing crop (e.g., point A to point B in Figure 1). If mixing shifts the controlling mechanism from nutrient limitation to light limitation, a moderate increase in mixed depth can cause a substantial rise of peak algal biomass or at best only a slight decline (A to C or B to C, respectively, in Figure 1). With large increases in mixed depth, the imposition of light limitation might cause substantial decreases in water column algal biomass (B to D in Figure 1). When algal biomass decreases with increased mixed depth the concentration of algae will decrease dramatically because less biomass is distributed in a much larger water volume. Finally, because of differences in growth parameters among algal species, a major shift in species composition could generate a change in peak quantity of algae apart from the effects of mixed depth.

In oligotrophic lakes, artificial destratification usually produces little change in cell concentrations (Knoppert, et al. 1970; Biederman and Fulton, 1971; Toetz, 1977a, b; but see Fast, 1971a). Sometimes, standing stock increases due to change in mixing depth, although the change is small in cases of incomplete destratification. Since the slope of the ascending curve in Figure 1 equals the peak nutrient-limited concentration of algae (Lorenzen and Mitchell, 1975), the slope will be smallest for oligotrophic lakes. Hence, any given change in mixed depth over the range of nutrient-limited biomasses will result in only small displacements of standing crop in oligotrophic lakes compared with potential shifts in richer lakes (also, see Forsberg and Shapiro in this volume on shifts in peak biomass with changing total phosphorus levels).

Blue-green species often control their depth distribution via buoyancy regulation to take advantage of specific optima in light, temperature, and nutrients (Fogg and Walsby, 1971; Konopka, et al. 1978). Artificial circulation disperses metalimnetic populations and causes overall decline of *Oscillatoria* spp.(Bernhardt, 1967; Weiss and Breedlove, 1973). Whatever the mechanism, *Anabaena* spp. are among the most sensitive forms (Ridley, 1970; Knoppert, et al. 1970; Malueg, et al. 1971; Steichen, et al. 1974; Barnett, 1975).

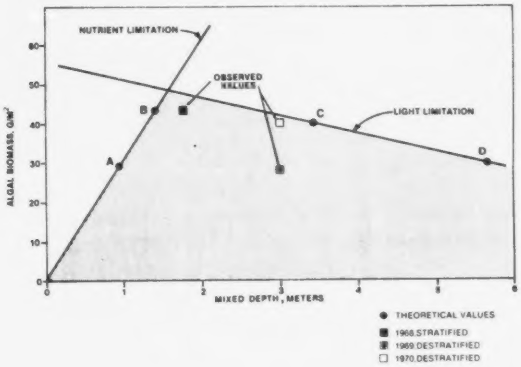

Figure 1. — Theoretical and observed peak biomass of algae in Kezar Lake (Lorenzen and Mitchell, 1975).

Chemical Mechanisms

Mixing techniques have been applied to reduce algal blooms by curtailing recycling of nutrients from the profundal zone (Toetz, et al. 1972; Dunst, et al. 1974; Fast, 1979a). Although PO concentrations are indeed lowered by destratification (e.g., Haynes, 1973; Toetz, 1979), the flux of nutrients from profundal sediments to the overlying water and subsequent uptake by the plant community could actually increase. Under aerobic conditions, the higher temperatures in the sediments after destratification will stimulate decomposition and release of phosphorus to overlying waters (Hargrave, 1969; Fast, 1971a; Kamp-Nielsen, 1975). Simultaneously, nutrient exchange across the mud-water interface is facilitated by increased flow of water over the sediments and invasion of burrowing macroorganisms which mix the sediments vertically, e.g., chironomid larvae (Porcella, et al. 1970; Gallepp, et al. 1978). Lastly, it is unlikely that circulation techniques can reduce internal loading of nutrients from other sources such as "leakage" from littoral macrophytes (Demarte and Hartman, 1974; Lehman and Sandgren, 1978).

Even if artificial circulation does reduce phosphorus regeneration from the sediments, significant changes in the biota will occur only if internal loading of nutrients is large relative to input from the watershed and algal growth is limited by phosphorus (Fast, 1975). Although the latter appears true in many instances (Likens, 1972), lakes with nuisance algal blooms are usually eutrophic and, by definition, experience high external loading. Also, Lane and Levins (1977) caution against overreliance on the concept of a single limiting nutrient.

The effects of destratification on the concentrations of dissolved inorganic nutrients in the upper waters of a lake are unpredictable due to interactions with biota and organic factions (Toetz, et al. 1972; Fast, 1975). For example, nutrients may be released by lysed algae (Robinson, et al. 1969; R. S. Kerr Res. Center, 1970; McNail, 1971), by decomposition of resuspended detritus (Hooper, et al. 1953; Fast, 1971a; Haynes, 1973), or by an abundant zooplankton population (Devol, 1979).

Biological Mechanisms

An effective destratification often causes a dramatic shift in species composition of the phytoplankton community, from dominance by one or a few species of blue-greens to a predominant assemblage of green algae (Table 1). Zooplankton readily graze on green algal species, whereas they reject the inedible and sometimes toxic blue-greens or grow poorly on them (Arnold, 1971; Porter, 1973; Webster and Peters, 1978). On the other hand, some gelatinous greens actually profit from passage through the gut of a *Daphnia* (Porter, 1975).

Shapiro (1973; Shapiro, et al. 1975) has induced the blue-green to green shift in experimental enclosures by adding CO_2 or HCl, both of which lower the pH of the water. Moreover, adding NO^8 and PO_4 facilitates the shift. Since the blue-greens decline precipitously before the greens begin growing rapidly, Shapiro, et al. (1975) suggest that the shift is mediated by the action of cyanophages (Shilo, 1971; Lindmark *in* Shapiro, 1979), rather than by direct competitive replacement. Indeed, the release of large quantities of $PO_4^{\equiv}$ and NH to the water after the sudden decline of blue-greens in the enclosures suggests that lysis is occurring.

Destratification essentially mimics Shapiro's experimental treatments by adding CO_2 and nutrients to the surface waters through: (1) Mixing of hypolimnetic CO_2 and nutrients into the surface layer; (2) recarbonation of waters by atmospheric exchange; and (3) decreasing the ratio of primary production to respiration through deepening of the mixed layer. In experimental enclosures, a change in algal species composition occurs only at pH values less than 8.5, and the results are unpredictable between pH 7.5 and 8.5 (Shapiro, et al. 1975). Lakes where pH decreased following circulation also showed an increase in the ratio of green to blue-green algae; whereas experiments that failed to lower the pH also failed to produce the shift to greens (Table 2).

In Kezar Lake during 1969, mixing caused a temporary rise in pH, but after 20 days of aeration, the pH dropped from 9.0 to 7.1, and at least a small increase in the ratio of greens to blue-greens ensued (N.H. Water Supply Pollut. Control Comm. 1971). Destratification by pumping hypolimnetic water to the surface maintained relatively low pH in the epilimnia of four Ohio lakes and prevented the usual fall blooms of blue-green algae (Irwin, et al. 1966).

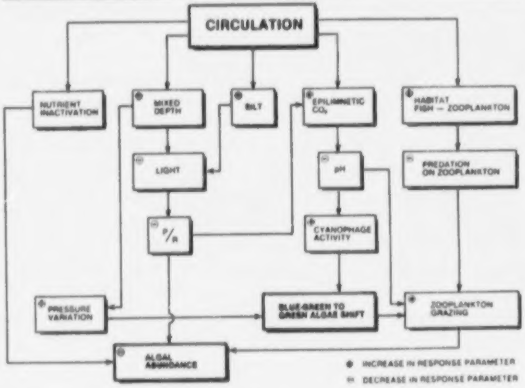

Figure 2. — Beneficial effects of artificial circulation on phytoplankton (Shapiro, 1979).

Table 2. — Epilimnetic pH changes associated with artificial circulation.

Lake	Reference	Direction of Change	pH Values Before	pH Values After
Group I[a]				
Cline's Pond	Malueg, et al. 1971	-	6.2-9.6[c]	6.4-7.2
University Lake	Weiss and Breedlove, 1973	-	7.6[d]	7.3, 7.0
	N.H.W.S.P.C.C. 1971 Haynes, 1973	1968 -	9.4	6.7
Kezar Lake	N.H.W.S.P.C.C. 1971 Haynes, 1973	1969 +	6.6	9.0
Stewart Hollow	Irwin, et al. 1966	-	6.8	5.5
	Irwin, et al. 1966	-	6.8	6.5
Caldwell Lake	Irwin, et al. 1966	0	7.3	7.0-7.5
Pine Lake	Irwin, et al. 1966	0	6.9-7.2	6.7-7.1
Vesuvius Lake	Irwin, et al. 1966	-	6.8-7.3	6.8-7.0
Buchanan Lake	Brown, et al. 1971	-	7.1	6.7
Group II[b]				
Parvin Lake	Lackey, 1972	0	6.6-7.2[d]	6.7-7.2
Test Res. I & II	Knoppert, et al. 1970	0?	?	>9.0
Starodworski Lake	Lossow, et al. 1975	-	9.0-9.4[d]	7.3-8.6
Lake Calhoun	Shapiro and Pfannkuch, 1973	0	8.0-8.5[d]	8.0-8.5
Ham's Lake	Steichen, et al. 1974	1973 - 1975	8.5	7.5
	Toetz, 1977b	0	>8.0	>8.0
Arbuckle	Toetz, 1977b	1975 - 1977	7.71[d]	7.39
	Toetz, 1979	0	~7.5[d]	~7.5
Lake Catharine	Kothandaraman, et al. 1979	0	>8.0[d]	>8.0
Hyrum Res.	Drury, et al. 1975	±	7.8-8.9[d]	7.2-9.2
El Capitan Res.	Fast, 1968	0	7.5-8.6[d]	7.7-8.3

[a] Group I = Lakes in which the ratio of green algae to blue-green algae increased after treatment
[b] Group II = Lakes in which the ratio of green algae to blue-green algae decreased or stayed the same after treatment
[c] Control section
[d] Control year, summer values

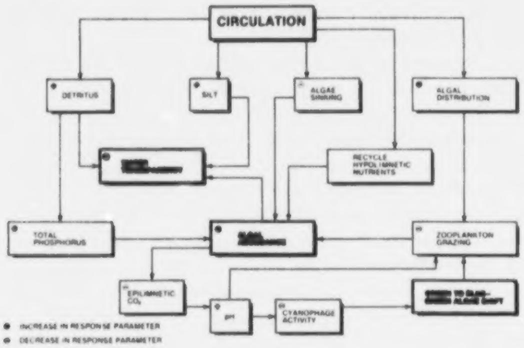

Figure 3. — Some adverse impacts of artificial circulation and their role in promoting blue-green algae blooms (Shapiro, 1979).

Although mixing caused a temporary decrease of epilimnetic pH in Ham's Lake (1973 experiment) and Starodworski Lake (Poland), the pH remained above 7.3 in both cases, failing to produce a shift from blue-green algae to green algae (Tables 1 and 2). In Hyrum Reservoir, where aeration caused microstratification and a reduction in mixed depth, pH of the surface waters rose sharply to 9.2 during a bloom of *Aphanizomenon* (Drury, et al. 1975).

Figures 2 and 3 summarizes some of the important mechanisms underlying the effects of artificial circulation on phytoplankton. The risk of adverse impacts can be minimized by proper design and application of the mixing system (see Pastorok, et al. In press).

EFFECTS OF ARTIFICIAL CIRCULATION ON ZOOPLANKTON AND SPECIES INTERACTIONS IN OPEN WATER

Artificial circulation generally leads to an increase in the abundance of zooplankton and an expansion of their vertical distribution (Table 3). Several studies reported no effects of mixing on the zooplankton but this result is probably due to inadequate sampling design (Eufaula Reservoir), incomplete mixing (Hyrum Reservoir, Arbuckle Lake), or lack of control data (Ham's Lake, Arbuckle Lake).

Depth Distribution

Most investigators have observed profound changes in distribution of cladocerans (e.g., Shapiro, et al. 1975; Brynildson and Serns, 1977). Lackey (1973b) found that the depth distributions of Cladocera and rotifers were generally unaffected by artificial circulation, but *Diaptomus* spp. tended to occur in deeper water during the treatment year. Even in lakes where zooplankton occupy the entire water column before treatment (e.g., Ham's Lake and Starodworski Lake), circulation usually shifts the vertical profile of the population toward lower depths.

Zooplankton Abundance

Brynildson and Serns (1977) documented a fourfold increase in *Daphnia* spp. after mixing of Mirror Lake in September 1974. Although the density of small cladocerans including *Bosmina longirostris* and *Diaphanosoma leuchtenbergianum* showed no significant

Table 3. — Responses of zooplankton to artificial circulation[a].

Lake	Reference	Abundance[b]	Depth Distribution	Ratio Copepods: Cladocerans
Buchanan Lake	Brown, et al. 1971	+	+	
Lake Roberts	McNall, 1971	+		
Lake Calhoun[e]	Shapiro and Pfannkuch, 1973	+	+	-
Stewart Lake	Barnes and Griswold, 1975	1974 - 1975		
Indian Brook Reservoir	Riddick, 1957	+	+	
Mirror Lake	Brynildson and Serns, 1977	1973 +	+	+
		1974 +	+	-
Parvin Lake	Lackey, 1973b	-	+	+
El Capitan Reservoir	Fast, 1971b	+	+	
Starodworski Lake[c]	Lossow, et al. 1975	+	0	
Eufaula Reservoir[d,e]	Bowles, 1972	0	0	0
Ham's Lake[c]	McClintock, 1976	0	0	0
Arbuckle Lake[c,e]	McClintock, 1976	0	0	0
Hyrum Reservoir[d,e]	Drury, et al. 1975	0	0	0

[a] + = decrease, 0 = no significant change
[b] Weighted mean density or standing stock
[c] Zooplankton distributed to bottom before mix
[d] Inadequate sampling design or lost samples
[e] Incomplete mix

change after circulation, calanoid and cyclopoid copepods increased during both experiments.

During aeration of Starodworski Lake, the relatively large *Daphnia hyalina* appeared for the first time and became especially abundant in lower water (Lossow, et al. 1975). *Bosmina longirostris* declined to particularly low densities during summer of the treatment and was replaced by the larger *B. coregoni*. *Chaoborus* larvae which are significant predators on small zooplankton (cf. Pastorok, in press), declined during aeration, relieving the predation pressure on *Bosmina*.

Shapiro, et al. (1975) found that the abundance of *Daphnia* spp. increased five to eight times during artificial circulation of Lake Calhoun compared with the previous control year. Moreover, the large-bodied *D. pulex* invaded the lake and became reasonably common after treatment. Other zooplankters, including cyclopoids, *Diaptomus*, *Bosmina*, and *Diaphanasoma* increased less. Although Lackey (1973b) reported a significant decline in the population of *D. schodleri* and Cladocera in general during treatment of Parvin Lake, the control year may have been unusual due to absence of the late summer bloom of *Aphanizomenon flos-aquae* (cf. Lackey, 1973a).

The growth of zooplankton populations following destratification could be caused by several factors: (1) Resuspension of detritus, creating additional food resources for filter-feeders (Saunders, 1972); (2) the shift from blue-green algae to green algae (Table 1); and (3) habitat expansion for both zooplankton and planktivorous fishes (Table 3). The dimly lit bottom waters serve as a refuge for zooplankton, protecting them from visual predators (Zaret and Suffern, 1976). The reduction in encounter rate between fish and their prey lessons predation pressure on the zooplankton, allowing population growth and invasion of large-bodied forms, especially *Daphnia* (Shapiro, 1979; cf. Hrbacek, et al. 1961; Andersson, et al. 1978; DeBernardi and Guissani, 1978).

In turn, large herbivores such as *Daphnia pulicaria* are more effective grazers of algae than are small zooplankton (Haney, 1973; Hrbacek, et al. 1978). They also release less phosphorus per unit body weight than the smaller forms (Bartell and Kitchell, 1978). Andersson, et al. (1978) found that dense populations of fish in experimental enclosures resulted in low numbers of planktonic cladocerans, high concentrations of chlorophyll, and blooms of blue-green algae. In enclosures without fish, large cladocerans prospered and grazed the phytoplankton down to low levels.

EFFECTS OF ARTIFICIAL CIRCULATION ON BENTHIC MACROINVERTEBRATES

The responses of benthic communities to lake aeration/circulation have been relatively consistent; i.e., increases in number of taxa, diversity, and biomass, especially in profundal areas (Table 4). In two lakes receiving low nutrient inputs, Parvin Lake and Section Four Lake, population densities showed a generalized decline or no change. Although the hypolimnion of Parvin Lake was normally anoxic during late summer while the deeper areas of Section Four Lake remained high in oxygen, the mechanisms producing declines in chironomid densities may have been similar. Both lakes normally had dominant chironomid assemblages in deep water prior to aeration. The decline in overall densities may have resulted from increased midge emergence due to the warmer bottom temperatures during lake circulation. In both lakes, the other insect larvae and invertebrates such as *Asellus* and *Hyalella*, which were abundant in littoral areas, did not invade the hypolimnion following aeration. Therefore, overall profundal biomass declined.

Four of the five lakes in which *Chaoborus* formed a significant component of the profundal benthos displayed a general decline in larval density following aeration (Table 4). The exception was Parvin Lake in which *Chaoborus* density did not change. In Cox Hollow Lake there was pronounced decline in *Chaoborus* associated with replacement of *C. punctipennis* by *C. albatus* (Wirth, et al. 1970). Prior to aeration *Chaoborus* was the only profundal macroinvertebrate in Stewart Lake, but following treatment the larvae were almost completely absent, having been replaced by oligochaetes and chironomids (Barnes and Griswold, 1975).

The distributional characteristics of *Chaoborus* are consistent with the observed declines in *Chaoborus* densities during lake aeration. Aeration of bottom strata removes the anoxic refugia of *Chaoborus*, thus exposing the larvae to intense fish predation. Since third and fourth instar *Chaoborus* are relatively large organisms (6 to 15 mm), they are a preferred food item for zooplanktivorous fish (Northcote, et al. 1978; von Ende, 1979). Field studies have shown that the migratory *C. punctipennis* occurs in lakes with fish while the non-migratory *C. americanus* is excluded from fish lakes (von Ende, 1979). Moreover, introduction of fish predators into lakes has virtually eliminated *C. americanus* and markedly reduced the densities of *C. trivittatus*, a deeper dwelling species (Northcote, et al. 1978).

In lakes showing declines in *Chaoborus* densities during aeration, the profundal areas were occupied by increased densities of other fauna such as oligochaetes, chironomids, and other insect larvae (e.g., Wilhm and McClintock, 1978. Sikorowa, 1978). These detritivores responded to the generally rich deposits of organic material by establishing relatively high standing crops. Thus, aeration may modify the trophic structure of the community by reducing zooplankton predators (i.e., *Chaoborus*) and increasing benthic detritivores. Organisms such as chironomid larvae are important food items for a variety of fish species. The high utilization of benthic fauna and the influence of fish predation on prey population densities are indicated in field studies such as Andersson, et al. (1978).

EFFECTS OF ARTIFICIAL CIRCULATION ON FISHES

In stratified lakes, coldwater fishes such as salmonids may be compressed into a narrow layer of available metalimnetic habitat by warm water above and anoxic conditions below. In all cases where depth distribution has been evaluated, fish have been observed to expand their vertical distribution in response to lake destratification.

Prior to destratification of Mirror Lake, trout and yellow perch were confined to the epilimnion and metalimnion (Brynildson and Serns, 1977). During the spring and late summer the maximum depth occurrence of the two fish species was about 5 and 7 meters, respectively, corresponding to a dissolved oxygen level of about 3 to 4 mg/l. After destratification fish were distributed throughout the water column to the maximum depth of 13 meters. Trout were essentially evenly distributed while yellow perch occurred from 4 to 13 meters.

After partial destratification of Lake Arbuckle in late summer, gizzard shad, freshwater drum, white crappie, and black bullhead all displayed increased depth distributions when compared with pre-circulation conditions (Gebhart and Summerfelt, 1976). In 1975 the total available fish habitat (as defined by the 2 mg/l DO isopleth) increased from 53 percent of lake volume in August to 99 percent of total volume in September following treatment. Habitat expansion has also been observed in El Capitan Reservoir for channel catfish, threadfin shad, and walleye (Fast, 1968), and in Lake Calhoun for yellow perch, bluegill, and crappie (Shapiro and Pfannkuch, 1973). Aeration of Casitas Reservoir has allowed the establishment of a year-round trout fishery (Barnett, 1975).

It is generally assumed that an expanded habitat benefits fish populations because of increased food supply and alleviation of crowding into epilimnetic strata during the summer. Comprehensive studies at Lake Arbuckle did indicate increased growth of bottom-feeding fishes, but the results varied with species and year of study (Gebhart and Clady, 1977). Increased growth at Stewart Lake (Barnes and Griswold, 1975) was apparently caused by selective elimination of stunted bluegills. Although the stimulation of fish growth and production is a conceivable benefit of aeration, it has not been evaluated in most projects. Most studies were of limited duration and fish populations may not have reached equilibrium with the modified lake environment. Moreover, some of the lakes contained already stressed populations (e.g., overcrowded and stunted centrarchids) slow to respond to habitat improvements (e.g., Wirth, et al. 1970).

In lakes with severe winter-kill problems, aeration during fall and winter reduces mortality rates (Halsey, 1968). In warmer areas, circulation of the lake in summer will increase the heat budget and may result in adverse water temperatures for salmonids, such as occurred in Puddingstone Reservoir (Fast and St. Amant, 1971). Localized aeration and partial destratification could allow for some cooler areas with sufficient DO for trout survival (e.g., Casitas Reservoir); however, the potential for other benefits such as water quality changes would be considerably less.

Adverse impacts follow destratification whenever the oxygen demand associated with resuspended particulates and reduced compounds lowers dissolved oxygen in the entire lake to levels below 2 to 3 mg/l. A dissolved oxygen concentration of at least 5 mg/l is generally required for maintenance of good game fish populations (U.S. EPA, 1976).

Table 4. — Responses of benthic macroinvertebrates to artificial circulation.

Lake	Reference	Organism density	No. of Species (or diversity)
Ham's Lake	Wilhm and McClintoch, 1978	+	+
Starodworskie Lake	Sikorowa, 1978	+	+
Lake Catherine	Kothandaraman, et al. 1979	+	+
Parvin Lake	Lackey, 1973c	varied[b]	0
El Capitan Reservoir	Inland Fish. Branch, 1970	+	+
Cox Hollow Lake	Wirth, et al. 1970		+
University Lake	Weiss and Breedlove, 1973	+[a]	+[a]
Stewart Lake	Barnes and Griswold, 1975		+
Section Four Lake	Fast, 1971a	-	

[a] Chironomids only

[b] Chironomids -, others 0

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PREDICTING THE ALGAL RESPONSE TO DESTRATIFICATION

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ABSTRACT

The response of phytoplankton communities to artificial destratification has been quite variable. The mechanisms underlying this variability were investigated in eight field experiments on two Minnesota lakes. Polyethylene enclosures were used in controlled experimental designs to investigate specific response mechanisms. A mathematical model was developed to describe the community response under different mixing regimes. The peak concentration and total amount of chlorophyll a in the mixed layer were predicted to either increase, decrease, or remain the same depending on changes in the mixed depth and the concentration of total phosphorus in the mixed layer following destratification. Changes in species composition during artificial circulation depended on the mixing rate achieved. Blue-green algae increased in relative abundance at the slower mixing rates while green algae and diatoms were favored at the fastest mixing rates. The shift to green algae occurred only during conditions of low pH and high nutrient availability associated with rapid mixing and is therefore most likely to occur when relatively deep productive lakes are rapidly mixed.

INTRODUCTION

Artificial circulation often has been proposed as a method for controlling algal blooms in lakes and reservoirs. However, in practice, the response of the phytoplankton to this treatment has been quite variable. Pastorok, et al. (1980) have recently summarized the results of a large number of destratification experiments. In 40 experiments where destratifcation was relatively complete, they found that only 65 percent led to a significant change in algal biomass; of these, 30 percent increased and 70 percent decreased algal biomass. Changes in algal species composition following destratification have also been variable. A shift in dominance from blue-green to green species has been reported by several investigators (Irwin, et al. 1966; Malueg, et al. 1971; Weiss and Breedlove, 1973). However, in some cases diatoms (Bernhardt, 1967) and in others blue-greens (Knoppert, et al. 1970; Drury, et al. 1975) have increased in relative abundance. These results indicate that the effects of artificial circulation on the phytoplankton are not always beneficial. It is therefore important that we understand the mechanisms underlying these effects so that our ability to predict the algal response will improve and circulation techniques can be used more effectively.

A number of mechanisms have been proposed to explain the response to the phytoplankton during artificial circulation. Several authors have constructed mathematical models of algal growth to predict the response at the community level (Murphy, 1962; Bella, 1970; Lorenzen and Mitchell, 1973; Oskam, 1978). While these models often ignore important aspects of algal growth (e.g. algal losses due to sinking and grazing), preliminary tests (Lorenzen and Mitchell, 1975; Oskam, 1978) indicate that this general approach may eventually provide a theoretical framework for predicting the community response. Fewer mechanisms have been proposed to explain shifts in species composition associated with artificial circulation. However, Shapiro (1973) has suggested that shifts from blue-green to green species reported in the literature might be related to decreases in pH and increases in nutrient availability which sometimes occur following destratification. He found similar shifts when he reduced pH and added nutrients to natural assemblages of algae in controlled field experiments. The fact that most of the blue-green to green shifts found during artificial circulation also occurred during conditions of low pH (Irwin, et al. 1966; Haynes, 1971; Weiss and Breedlove, 1973) tends to support this hypothesis.

We present here an overview of the results from eight field experiments which were conducted over a period of 3 years on two Minnesota Lakes. These experiments were designed to investigate specific mechanisms proposed to explain variability in the algal response during artificial circulation. Particular emphasis was placed on developing and testing a mathematical model capable of describing the community level response and on evaluating the pH-shift mechanism proposed by Shapiro (1973). Our purpose in this paper is to summarize those results which have a direct bearing on our ability to predict the algal response during artificial circulation.

STUDY SITES

The experiments were conducted on two small eutrophic lakes near Minneapolis and St. Paul during the ice-free months between 1976 and 1978. Little Lake Johannah, which has a surface area of 7.3 hectares and a maximum depth of 13 meters, was the site of experiments 1, 2, 4, 5 and 7. Experiments 3, 6 and 8 were carried out on Twin Lake which has a surface area of 15 hectares and a maximum depth of 12 meters. Additional details on the limnology of Twin Lake and Little Lake Johannah are described elsewhere (Allott, 1979; Shapiro, et al. in prep.).

EXPERIMENTAL DESIGNS AND METHODS

The experiments were designed to simulate the effects of mixing without circulating a whole lake. This was accomplished by enclosing vertical columns of lake water in polyethylene bags and then circulating with compressed air. These enclosures were made of 6 mil extruded polyethylene cylinders with a diameter of 1 meter and depth of 8 meters. They were open at the top, reinforced with PVC tubing on the sides, and either open or closed at the bottom. Open bottom enclosures were used to simulate natural conditions. They were held open at the bottom by weighted PVC hoops and lowered slowly from the surface to entrain an undisturbed column of water.

Closed bottom enclosures were used to study the effects of selected hypolimnetic constituents. These bags were first filled with surface water and then allowed to stratify through thermal conduction with their surroundings (the thermocline was generally at a depth of 3 meters in Little Lake Johannah and 5 meters in Twin Lake during the experiments). Water was then withdrawn from the hypolimnetic portions and, after specific additions, returned to the same depth. These additions included various combinations of nitrogen, phosphorus, alkalinity, and carbon dioxide designed to simulate natural hypolimnetic levels.

In each experiment several enclosures were suspended from outriggers attached to rafts as shown in Figure 1. Various treatments were then applied to the different enclosures in controlled experimental designs to investigate specific response mechanisms. In addition to the manipulations of hypolimnetic chemistry in the closed bottom bags, the mixing rate and mixed depth were varied in the enclosures by adjusting the air flow rate and depth of air release, respectively. The flexibility of this general design made it possible to simulate a wide range of mixing conditions. For a detailed description of specific experimental designs and analytical procedures refer to Shapiro, et al., in prep.

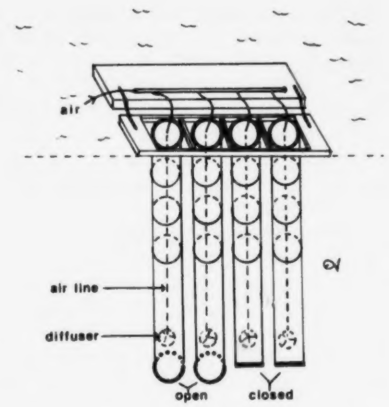

Figure 1. *In situ* apparatus for suspending experimental enclosures.

RESULTS AND DISCUSSION

By varying the mixing regime within the different experimental designs it was possible to produce a range of algal responses similar to that observed in whole lake destratification experiments. This made it possible to evaluate specific response mechanisms proposed to explain this variability in the algal response. Mechanisms of potential importance at both the community and species level were evaluated.

The algal community response during artificial circulation. A mathematical model of algal growth was constructed to provide an appropriate theoretical framework for evaluating the community response during artificial circulation. (refer to Forsberg and Shapiro, 1980, and Shapiro, et al. in prep. for a more detailed development of the model) While it contains elements of expressions presented by Talling (1957), Megard (1974), and Senft (1978) the development of the model follows directly from the work of Lorenzen and Mitchell (1973). They considered the effects of nutrient and light limitation independently and developed separate expressions to predict nutrient and light limited peak algal biomass following destratification. We also evaluated the effects of both nutrient and light limitation on algal growth, but, instead of treating them separately, we considered their effects simultaneously and derived a single expression for the peak concentration of chlorophyll *a*

$$c^* = \frac{\mathrm{Ln}(I_o/I_{z'})P_{sat} - D\Theta Z_m e_w}{D\Theta z_m e_c + [\mathrm{Ln}(zI_o/I_{z'})P_{sat}k_q]/TP}$$

where,

c^* = the peak concentration of chlorophyll *a* in the mixed layer (mg Chl m^{-2})
I_o = the light intensity just below the surface
$I_{z'}$ = the light intensity at the depth z'
z' = the depth empirically defined as, z' = (daily integral rate of photosynthesis, mg C $m^{-2}d^{-1}$)/

(maximum daily volumetric rate of photosynthesis, mg C $m^{-3}d^{-1}$)
P_{sat} = the maximum daily specific rate of photosynthesis in a nutrient saturated mixed layer (mg C mg $Chl^{-1}d^{-1}$)
D = the specific loss rate (day^{-1})
Θ = the ratio of carbon to chlorophyll *a* in the algae mg C mg Chl^{-1})
z_m = the depth of the mixed layer (meters)
e_c = the partial attenuation coefficient of chlorophyll *a* (m^2 mg Chl^{-1})
e_w = the residual extinction coefficient of the water (m^{-1})
k_q = the level which the ratio of total phosphorus to chlorophyll *a* must exceed before photosynthesis will occur (mg P mg Chl^{-1})
TP = the concentration of total phosphorus in the mixed layer (mg P m^{-3})

Data from 10 experimental enclosures in experiment 6 (Twin Lake) were used to evaluate the parameters in equation 1 and provide a preliminary test of the model. The enclosures were all closed at the bottom. Nitrogen, phosphorus, and alkalinity were added to the hypolimnia of circulated enclosures before mixing in amounts designed to simulate natural levels. Eight of the enclosures were artificially circulated to a depth of 7 meters while two control bags remained stratified with a mixed depth of 5 meters.

TP and z_m were the parameters in equation 1 which changed significantly during artificial circulation. All other factors were assigned constant values. The model was then used to simulate the effects of changes in TP and z_m on the peak concentration of chlorophyll *a* in Twin Lake. The results of this simulation are shown in Figure 2a. Each line in this figure indicates the effect of changes in z_m on c* at a single level of TP. The TP levels chosen for the simulation were those observed in the lake and in several of the enclosures at the point of maximum yield (i.e., maximum observed ration, chlorophyll/total phosphorus). The chlorophyll concentrations observed at the point of maximum yield provided field estimates of c* and are indicated by black circles on Figure 2a at the appropriate mixed depths. The lines connecting these black circles to the simulation lines represent the differences between the predicted and observed c* values.

These differences are generally small indicating good agreement between the simulation and field results. The simulation results indicate that, at a given level of TP, c* will decrease as the mixed depth is increased. However, if the concentration of TP in the mixed layer changes during mixing, c* may either increase, decrease, or remain the same depending on the direction and magnitude of the change. When the mixed depth was increased from 5 to 7 meters in the circulated enclosures the concentration of TP increased in the mixed layer changes during mixing, c* may either increase, decrease, or remain the same depending on the direction and magnitude of the change. When the mixed depth was increased from 5 to 7 meters in the circulated enclosures the concentration of TP increased dramatically, resulting in a large increase in the concentration of chlorophyll *a* (c.f. Figure 2a). Because of this increase in TP, the mixed depth would have to be increased to a depth greater than 20 meters before a reduction c* would occur. Since the mean depth of Twin Lake is only 5.5 meters, destratification would not be effective in reducing the peak chlorophyll concentration.

The model was also used to simulate the effects of changes in TP and z_m on peak algal biomass which was defined as the total amount of chlorophyll *a* beneath a square meter of lake surface or c^*z_m. The results of this simulation are shown in Figure 2b. Again, the circles represent field observation and the agreement between predicted and observed results is good. The model predicts that, at a given level of TP, peak biomass will reach a maximum level at a mixed depth of about 15 meters. Peak biomass will increase during artificial circulation if the final mixed depth is less than this value and may either increase, decrease or remain the same at greater mixed depths. However, if the concentration of TP increases during mixing, as it did in the circulated enclosures, the level of c^*z_m achieved will be higher than would otherwise be expected.

Peak biomass was much higher in the circulated enclosures than in either the lake or the control bag and this difference was primarily due to the increase in TP which occurred during mixing. These results indicate that, even if TP didn't change, Twin Lake would have to be mixed to a depth greater than 30 meters (about six times its mean depth) before a reduction in peak biomass would occur.

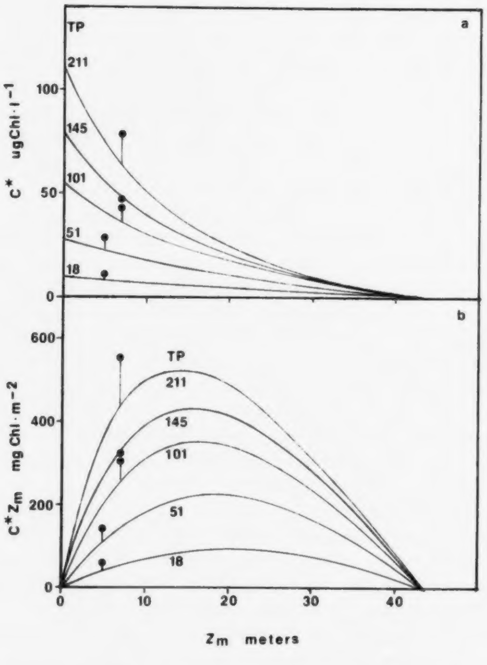

Figure 2.–The effect of changes in the mixed depth, z_m, and the concentration of total phosphorus, TP (mg P m^{-3}), on (a) the peak concentration of chlorophyll *a* c*, and (b) the peak algal biomass, c^*z_m in Twin Lake. The black circles represent field observations from experiment 6. The TP levels of 18 and 51 represent values for the control enclosure and lake, respectively.

Lorenzen and Mitchell (1975) presented a similar analysis for Kezar Lake, N. H. Although they used a different model which did not consider the effects of changes in nutrient concentrations, the results were qualitatively the same. They also predicted that peak algal biomass would reach a maximum at a particular mixed depth, which they defined earlier (1973) as the optimum mixed depth, z_{opt}. However the value of z_{opt} which they predicted for Kezar Lake (about 1.5 meters) was much lower than the value found for Twin Lake (15 meter). The lower value for z_{opt} in Kezar Lake means that a much smaller increase in z_m would be required to reduce peak biomass. This difference in z_{opt} between lakes is apparantly due to differences in the response characteristics of the two communities involved.

The results of the simulation for Twin Lake suggest that increases in TP which often occur during artificial circulation can significantly affect community response and, in some cases, can increase the concentration of chlorophyll *a*. Total phosphorus concentrations were found to increase in the circulated enclosures in all eight of the experiments conducted on Twin Lake and Little Lake Johannah. These increases in TP generally resulted in higher concentration of chlorophyll *a*. In six of the eight experiments, a significant positive relationship was found between the average concentrations of TP and chlorophyll *a* determined in the enclosures. The regression lines from these relationships are shown in Figure 3. The regression lines for different experiments in the same lake were generally similar (experiments 3, 6, and 8 were in Twin Lake; experiments 1, 2 and 5 were in Johannah). However, the response for each lake was quite different. Increases in TP generally resulted in much larger increases in chlorophyll *a* in Twin Lake than in Little Lake Johannah.

This difference in community response is apparently related to differences in the relative availability of nitrogen and phosphorus in the two lakes. The ratio of inorganic nitrogen to inorganic phosphorus, IN/IP, determined initially in each experiment is indicated on Figure 3. The IN/IP ratios were always greater than 30 in Twin Lake and less than 10 in Johannah. Forsberg, et al. (1978) surveyed a large number of lakes and found that above an IN/IP ratio of 12 most phytoplankton were P-limited, below a value of 5 they were generally N-limited, and between 5 and 12 either nutrient could limit algal growth. It is clear, then, that the phytoplankton in Twin Lake were limited by phosphorus while the low IN/IP and weaker community response found in Johannah suggest that the phytoplankton there were probably limited by nitrogen. These results indicate that, during circulation, the phytoplankton will respond to changes in the concentration of that nutrient which is in the shortest supply.

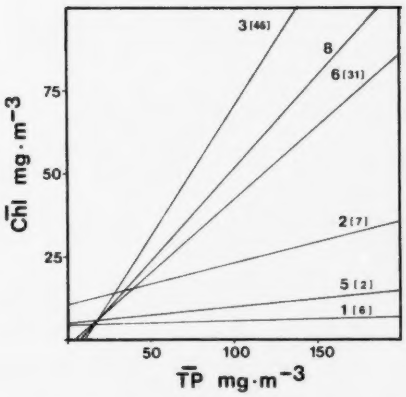

Figure 3.–The relationships found between average concentrations of total phosphorus and chlorophyll *a* in experimental enclosures for circulation experiments in Little Lake Johannah (1, 2 and 5) and in Twin Lake (3, 6 and 8). Numbers in parentheses indicate the ratios of IN/IP determined initially in each experiment.

Changes in Species Composition during Artificial Circulation. Changes in species composition during artificial circulation were found to depend primarily on the mixing rate. This effect was most apparent in the open bottomed enclosures. When these enclosures were mixed slowly surface levels of TP and pH generally increased. These conditions often increased the relative abundance of blue-green species such as *Anabaena circulinus* and *Microcystis aureginosis*. At the faster mixing rates, where complete chemical destratification occurred, larger increases in TP and nutrient availability were usually observed at the surface. In addition, increases in the concentration of CO_2, which occurred as hypolimnetic water was brought rapidly to the surface, generally resulted in lowered pH levels. These conditions often increased the relative abundance of green algae and diatoms. The green algae: *Sphaerocystis Schroederi*, *Ankistrodesmus falcatus*, and *Scenedesmus spp.*, grew particularly well at these faster mixing rates as did the diatoms: *Nitszchia spp.*, *Synedra spp.* and *Melosira spp.*

There was some evidence that reduced sinking losses may have given the diatoms an advantage at the faster mixing rates. This was demonstrated in experiments 1 and 2 where several enclosures were mixed rapidly but without increasing the mixed depth below the thermocline. This allowed us to separate the direct effect of turbulence from other factors which might change if the mixed depth were increased. The growth rates of diatoms were found to increase to much higher levels than those of green and blue-green species as the level of turbulence increased. The conditions of high nutrient availability and low pH which prevailed at the faster mixing rates are similar to those which Shapiro (1973) found to produce a shift from blue-green to green species in his field experiments. The fact that similar shifts to green species also occurred in the rapidly mixed enclosures suggests that a common mechanism might be involved.

Shapiro found that the growth of blue-green species was suppressed at low pH. We also observed a decline in the growth rates of blue-green algae as the pH dropped in the rapidly mixed enclosures. While the mechanism is not entirely clear, this disadvantage of blue-greens at low pH may involve the activity of cyanophage, Lindmark (1979) demonstrated dramatic increases in the incidence of cyanophage infection as the pH was lowered in laboratory cultures of blue-green algae.

Alternatively, King (1970) has suggested that shifts from blue-green to green species which he observed in sewage lagoons at low pH might be due to differences in carbon uptake kinetics between these two divisions. Long (1979) provided considerable support for this hypothesis. In a series of carbon growth and uptake experiments for a large number of species he demonsrated the general competitive superiority of green algae over blue-greens at low pH. He also showed how this competitive advantage shifts in favor of the blue-green species at high pH levels. This latter result may explain the dominance of blue-green algae at high pH in the slow mixed enclosures.

The shift from blue-greens to greens was not always observed in the rapidly mixed enclosures and apparently depended on the magnitude of the pH drop during mixing. It occurred most often in Johannah where relatively low alkalinity and high levels of hypolimnetic CO_2 resulted in a large drop in pH during mixing. The shift was seldom seen in Twin Lake where pH dropped only slightly during mixing due to much higher levels of alkalinity and lower hypolimnetic levels of CO_2.

SUMMARY

The results presented here suggest that the response of a particular phytoplankton assemblage during artificial circulation will depend on a number of factors. The response at the community level is apparently a complex function of many different lake and community characteristics which can only be described within a theoretical framework which considers all of these factors simultaneously. The model presented here represents an improvement over earlier attempts to provide such a framework.

Previous expressions did not consider the effects which changes in total nutrient concentrations might have on the phytoplankton. The results from the field experiments (Fig. 3) indicate that these changes can have a significant effect on the community response. By considering the effects of nutrient and light limitation simultaneously a single expression was derived which could be used to predict the level of algal biomass as a direct function of the mixed depth and total phosphorus concentration. The simulation results presented for Twin Lake demonstrated how both the total amount and concentration of chlorophyll *a* in the mixed layer could either increase, decrease, or remain the same depending on the changes in TP and z_m which occur during artificial circulation. A different approach may have to be taken in lakes such as Little Lake Johannah where algal growth was apparently limited by nitrogen instead of phosphorus.

However, it should be possible to develop a model, similar to the one presented here, which would predict the effects of changes in total nitrogen levels on the community response. Shifts in species composition during artificial circulation were found to depend primarily on the mixing rate achieved. Blue-green species were favored at the slowest mixing rates while greens and diatoms were favored at the faster mixing rates. While the mechanisms involved were not entirely clear, the shift in competitive advantage from blue-green species at the faster mixing rates was apparently related to the low pH and high nutrient levels achieved under these conditions. This shift would therefore be most likely to occur in relatively deep productive lakes with high hypolimnetic concentrations of CO_2 and nutrients where mixing will result in a large drop in pH and increases in nutrient level at the surface.

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RESERVOIR MIXING TECHNIQUES: RECENT EXPERIENCE IN THE UK

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ABSTRACT

The United Kingdom has about 180 impounding and pumped storage reservoirs of sufficient size and location which are likely to stratify thermally. Of these at least 30 have, or plan to install, a destratification system. The two systems most commonly used are submerged jetted inlets for pumped storage reservoirs and perforated-pipe air-mixing systems for impounding reservoirs. When operating, these systems maintain temperature differences between surface and bottom waters of 1 to 3 °C compared with differences of 8 to 9 °C under stratified conditions. The chemical quality of the water is also maintained at a higher standard.

INTRODUCTION

The United Kingdom has about 400 raw-water reservoirs having a capacity exceeding 20,000 m^3. Most of these are impounding reservoirs typical of upland storage while the remainder are either bunded pumped-storage systems, or pumped-storage impoundments, primarily situated in lowlands.

Approximately half the water consumed in the UK is supplied from storage reservoirs with direct river abstraction; ground water meets the remaining demand. The quality of reservoir water depends to a large extent on size, geographical location, the quality of the water used for refill, and the extent of thermal stratification.

Of the reservoirs in the UK, about 180 are likely to stratify in the spring and summer. Frequently associated with thermal stratification is hypolimnetic deoxygenation, and the poor chemical quality of this water severely restricts its use for water supply or river regulation. Excessive amounts of iron, manganese, and humic acids must be removed by water treatment and this may not always be achieved easily. Ammonia may interfere with sterilization by chlorine and possibly increase treatment costs. Sulfides cause unpleasant odors, demand much chlorine, and corrode iron and concrete.

For river flow regulation purposes this water is also undesirable. Most of the decomposition products contained in the water will exert an oxygen demand on the river when the water is used to augment the low summer flows. At such times the oxygen is most needed and the content is at its lowest. Sulfides and ammonia in sufficient concentrations are toxic to fish, especially in combination with low oxygen levels (Calif., State Water Qual. Control Board, 1963; Herbert, 1961). Precipitated iron oxide may coat macrophytes and settle on stream beds, consequently disturbing the stream ecosystem.

Two techniques have been used to mix reservoirs in the UK — inlet jetting and air injection. Both these methods achieve a high degree of mixing throughout the reservoir depth and in doing so maintain approximate isothermal conditions during the summer; this prevents chemical deterioration of water quality. Although artificial mixing is primarily intended as a method of improving the chemical quality of the water and making a greater volume of stored water available for use, it has been suggested (Steel, 1972; Lorenzen and Mitchell, 1975) that some control of algal populations is also achieved.

MIXING SYSTEMS USED IN THE UK

Two main directions have been followed in the UK to break down and prevent thermal stratification. The technique used for pumped-storage reservoirs involves pumping the incoming water through a nozzle situated near the bed of the reservoir, Figure 1. Water is entrained from the lower layers and this results in vertical and horizontal mixing which not only maintains the reservoir in an approximately isothermal state but ensures continual re-aeration at the surface. A diagrammatic representation of the induced circulation pattern is shown in Figure 2.

A technique used in impounding reservoirs is air injection, using either a confined or unconfined device. Confined-air injection consists of releasing compressed air through a series of vertical, free-standing polyethylene tubes positioned on the bed of the reservoir. The tubes are usually 2 to 3 meters long with a diameter of 0.3 to 0.4 meters. Two confined systems which have been employed over the last decade are the Aero-Hydraulics gun (Bryan, 1964) and the Helixor

Figure 1. — A normal and a jetted-inlet system in a pumped storage reservoir.

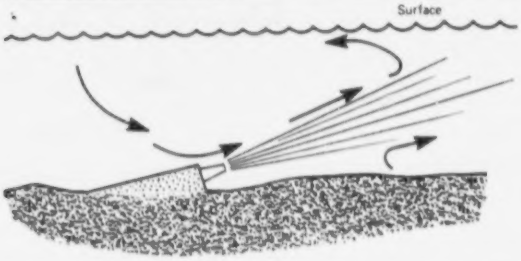

Figure 2. — A diagramatic representation of the induced circulation resulting from inlet jet mixing.

Figure 3. — A perforated-pipe system anchored by concrete blocks at the foot of an impounding dam.

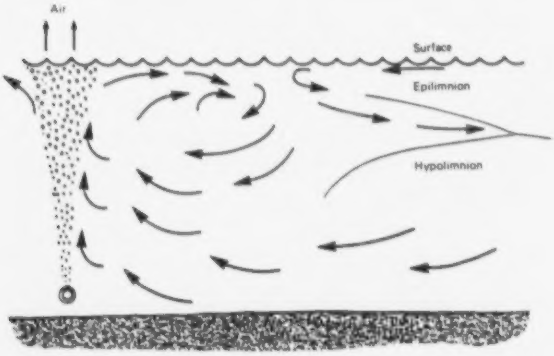

Figure 4. — Circulation of water induced by rising air bubbles from a perforated pipe.

(made by Polcon Corp., Montreal). In the former system a steady flow of large single air bubbles, which act like expendable pistons, force water through the tube, inducing circulation. The Helixor system, originally designed for wastewater treatment, is similar to the Aero-Hydraulics gun, but rather than a large single air bubble, the device relies on many smaller bubbles. These follow a spiral path within the tube itself entraining bottom water and inducing the necessary circulation.

Unconfined air injection has been used more commonly. In this system compressed air is pumped through a perforated pipe or a diffuser dome anchored near the bed of the reservoir. A perforated-pipe system usually has 100 to 200 meters of pipe with 0.8 mm diameter holes at 0.3 meter centers. An anchored pipe section is shown in Figure 3. Clean, oil-free air is supplied to the pipe by a compressor which may be sited some distance away. A diagrammatic representation of the circulation pattern induced by the rising bubbles is shown in Figure 4. Oxygen demands are met primarily through surface re-aeration.

A list of the reservoirs in which these systems have been installed, or are going to be installed, is shown in Table 1.

SUGGESTED DESIGN PROCEDURE FOR MIXING SYSTEMS

When considering the thermal behavior of water bodies, the year in the southern part of the UK, can be considered to be made up of four quarters. These are defined in Table 2 by the thermal effect and range of water temperatures associated with them. Clearly the critical time is the constant heating quarter, and any system designed to prevent stratification from becoming established must be designed to overcome the effects of this period.

Jetted-Inlet System

The objectives of a jetted inlet are:

1. To entrain water from the lower layers carrying it to, or near to, the surface where re-aeration takes place.

2. By their direction and momentum to circulate the general body of the water, giving an overall mixing effect.

The characteristics of a jet system may be identified by the densimetric Froude number (F) defined as

$$F = \frac{U}{\sqrt{gD\frac{\Delta\rho}{\rho}}} \qquad \text{eq. 1}$$

where,

U = mean jet velocity (ms^{-1})

g = acceleration due to gravity (ms^{-2})

D = diameter of jet nozzle (m)

$\Delta\rho$ = absolute density difference between pumped and ambient water (kgm^{-3})

ρ = density of pumped water (kgm^{-3})

As F tends to infinity, plume buoyancy is negligible and momentum dominates, while as F tends to zero momentum is negligible and the plume rises almost

Table 1. — Reservoirs in the UK having artificial mixing systems.

Reservoir	Volume $10^6 m^3$	Method of Filling	Mixing System
Kielder	204	Impounding	Perforated pipe*
Rutland Water	124	Pumped & Impounding	Helixor
Datchet	38	Pumped	Inlet & recirculation jets
Wraysbury	35	Pumped	Inlet & recirculation jets
Carsington	35	Impounding	Perforated pipe*
Bewl Bridge	31	Pumped & Impounding	Perforated pipe
Broad Oak	24	Pumped & Impounding	Perforated pipe*
King George VI	20	Pumped	Diffuser blocks
Wimbleball	20	Impounding	Perforated pipe
Queen Elizabeth II	20	Pumped	Inlet & recirculation jets
Hollowell	18	Impounding	Perforated pipe
Blithfield	18	Impounding	Perforated pipe
Loch Turret	18	Impounding	Air gun
Farmoor II	9	Pumped	Inlet & recirculation jets
Bough Beech	9	Pumped & Impounding	Recirculation jet
Blagdon	8	Impounding	Perforated pipe
Staunton Harold	7	Pumped & Impounding	Perforated pipe
Clatworthy	5	Impounding	Perforated pipe
Pitsford	5	Impounding	Perforated pipe
Farmoor I	5	Pumped	Inlet & recirculation jets
Upper Glendeven	5	Impounding	Air gun
Lower Glendevon	4	Impounding	Air gun
Castlehill	3	Impounding	Helixor
Cropston	3	Impounding	Perforated pipe
Ardleigh	2	Pumped & Impounding	Helixor & perforated pipe
Ravensthorpe	2	Impounding	Perforated pipe
Wistland Pound	2	Impounding	Perforated pipe
Hawkridge	1	Impounding	Perforated pipe
Lower Lliw	1	Impounding	Perforated pipe
Grimsbury	0.3	Impounding	Perforated pipe
Court Farm	0.3	Pumped	Perforated pipe

*Planned installation

immediately after leaving the nozzle.

Steel (1976) has reviewed the literature related to inlet jets and their characteristics. For a given nozzle diameter approximate relationships have been derived between jet trajectory, orientation, and densimetric Froude number.

$$Z = X \sec\theta \left\{ \sin\Theta + \frac{0.048}{F^2}\left(\frac{X}{D}\sec\theta\right)^2 \right\} ; 0 \leq \theta \leq 45^0 \qquad \text{eq. 2}$$

Where: Z is the height of the jet trajectory above the jet orifice (m)

X is the related distance from the jet orifice (m)

θ is the orientation of the jet to the horizontal

To meet the objectives stated earlier it is necessary to maximize trajectory length, thereby maximizing the total volume of water entrained, while ensuring that the entrained water reaches the surface layers. The choice of jet orientation and nozzle diameter will be a compromise to suit the range of operating conditions. This involves taking into account different inlet pumping rates and density differences between the inlet water and that in the reservoir. Indeed it may well be prudent, costs permitting, to have a choice of jet inlets of different diameter and orientation. One approach to designing such a system would be to determine the rate of energy required at the jet to maintain approximate isothermal conditions during spring and summer. For design purposes in temperate climates the proportion of solar radiation which appears as heat energy may be taken as approximately 5 J $m^{-2}d^{-1}$. In addition, the efficiency of energy transmission (η) associated with a jet system is about 2 to 5 percent which means that the rate of energy (E) required at the jet may be estimated by

$$E = \frac{5 \text{ x surface area}}{86{,}400 \times \eta} \quad (J\,s^{-1}) \qquad \text{eq. 3}$$

This energy rate may be related to mean nozzle velocity (U) for a range of inlet pumping rates (Qi, m^3s^{-1}) through,

$$E = 0.5\,\rho\,Qi\,U^2 \quad (J\,s^{-1}) \qquad \text{eq. 4}$$

where ρ is the density of the incoming water.

The mean nozzle velocity can then be determined from equation 4 as,

$$U = \left(\frac{2E}{\rho Qi}\right)^{\frac{1}{2}} \quad (m\,s^{-1}) \qquad \text{eq. 5}$$

The diameter of the nozzle follows from

$$D = \left(\frac{4Qi}{U\pi}\right)^{\frac{1}{2}} \quad (m) \qquad \text{eq. 6}$$

and the Froude number is calculated from equation 1.

The orientation of the jet may now be chosen using equation 2 so that for a range of operating conditions the design criteria are met.

Air Injection: Perforated-Pipe System

There are two main requirements of a perforated pipe destratification device.

1. After the onset of thermal stratification in spring or early summer the device must be capable of mixing the greater part of the reservoir volume in a reasonably short time (say 5 to 10 days depending on reservoir volume) so that approximate isothermal conditions exist over the depth.
2. During operation of the device the oxygen demands within the water column should be met initially through mixing between the upper and lower water layers and in the longer term through surface re-aeration.

A design procedure for this device (Davis, 1980) relies on assuming a design temperature profile corresponding to conditions in spring or early summer. Typically in the UK the design temperature profile will correspond to a 4°C temperature difference between the epilimnion and hypolimnion, with the temperature of the hypolimnion being around 8°C. From this the stability of the reservoir (the energy required to completely mix a stratified reservoir) is calculated. An estimate of the total energy required (E) to destratify the reservoir is obtained by adding to the stability value the solar heat energy input (approximately $5 \mathrm{Jm}^{-2}\mathrm{d}^{-1}$) during the destratification period.

An estimate of the free air required at the compressor can be obtained from the following relationship.

$$\frac{\text{Energy input by the perforated pipe to destratify reservoir}}{\text{Total theoretical energy required (E)}} = 20 \qquad \text{eq. 7}$$

The numerator in equation 7 is obtained by assuming isothermal conditions and a bubble pressure just sufficient to overcome the hydrostatic head. This is a function of the free air supplied by the compressor.

After re-arranging terms, equation 7 becomes

$$Q = \frac{0.196\,E}{T \ln(1 + H/10.4)} \quad (\mathrm{l\,s^{-1}}) \qquad \text{eq. 8}$$

Where: Q is the free air delivered
T is the time for destratification (s)
H is the depth of water above the pipe (m)

To calculate the length of perforated pipe required (L) the following relationship is used:

$$\frac{\text{Volume of water entrained by air bubbles to destratify reservoir}}{\text{volume of reservoir (V)}} = 2.5 \qquad \text{eq. 9}$$

The volume of water entrained for a given free air flow has been investigated by Bulson (1961); his empirical relationship is used.

Equation (9) then becomes

$$L = 3.73 \left\{ \frac{V^3\,[\,1 + H/10.4\,]}{T^3\,Q\,[\ln(1 + H/10.4)\,]^3} \right\}^{\frac{1}{2}} \quad (\mathrm{m}) \qquad \text{eq. 10}$$

The air pressure required at the compressor can now be calculated taking into account hydrostatic head and friction losses in the pipe work.

CASE STUDIES

The results from three different reservoirs employing inlet jetting, jetted recirculation, and perforated pipe air-mixing are presented. The morphometry of these reservoirs is given in Table 3.

Reservoir 1: Jetted-Inlet Systems

Water is pumped into the reservoir from the adjacent river to balance treatment plant demand and maintain the reservoir at top water level. River water is pumped in through a low-velocity 0.76 m diameter pipe or a high-velocity jetted inlet (0.3 m or 0.38 m diameter) inclined at 22.5°. Cost of jetting operation is approximately 2 percent over low velocity pumping cost.

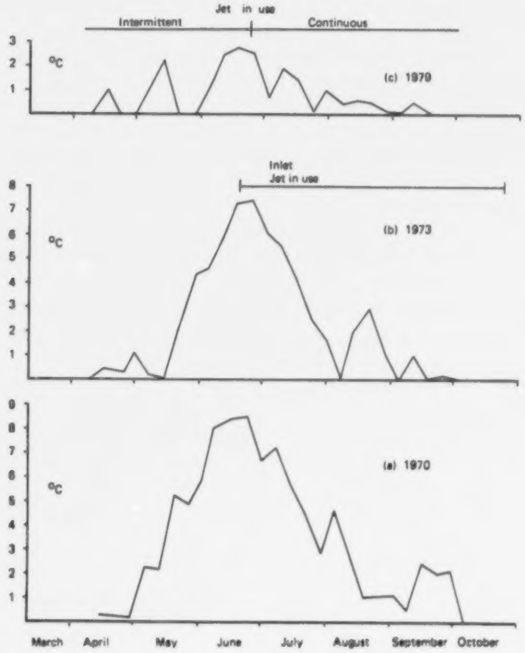

Figure 5. — Temperature differences between surface and bottom water (10 m) when (a) unjetted; (b) jetted after stratification was established; (c) jetted throughout the summer.

When the reservoir was first constructed the inlet pipes were not jetted and the water velocity at the inlet was not sufficient to prevent thermal stratification. Temperature differences of 8.5°C between surface and bottom water were observed (Figure 5a). One inlet pipe was then modified by fitting a reduced diameter nozzle inclined at 22.5° to the horizontal. In 1973 the reservoir was allowed to stratify. The inclined jet was then brought into use and the temperature differential decreased from 7.5 to <3°C (Figure 5b).

In subsequent years the system has been operated to prevent, as far as possible, any stratification taking place. Although temperature differences of up to 3°C (Figure 5c) have occurred, the bottom waters have not become anoxic, a minimum value of 50 percent of saturation being recorded.

Reservoir 2: Jetted Recirculation

Ninety percent of the stored volume of this reservoir is pumped from the river source between September and April; the only water entering the reservoir during the remainder of the year flows in from a natural feeder stream. Water is taken from the bottom of a draw-off tower near the dam wall and returned to the reservoir about 500 meters up the valley through a 0.46 m diameter jet inclined at 8° to the horizontal. A pumping rate of 0.53 $m^3 s^{-1}$ gives a mean nozzle velocity of 3.2 ms^{-1}. Pumping costs are currently estimated at £11 per day.

Filling of the reservoir began in 1969 to a depth of 12 meters. Summer stratification resulted in low dissolved oxygen levels with increases in dissolved iron, manganese, and ammonia. The following year the reservoir was at maximum depth by April. Summer stratification again resulted in low dissolved oxygen levels with increases in dissolved silica, phosphate, ammonia, and manganese, but not iron. The recirculation pumps were run during June and July for short periods and this smoothed the thermal profile although chemical stratification persisted. Natural overturn took place in October.

In subsequent years the reservoir stratified and as a result high levels of dissolved manganese occurred in the hypolimnion. The recirculation pumps were used intermittently to lessen the degree of stratification and decrease the levels of dissolved manganese. A typical pattern of events is shown in Figure 6, indicating the rapid improvement in water quality following operation of the recirculation system.

Reservoir 3: Perforated-Pipe System

The water supply to this reservoir is entirely from natural feeder streams. This reservoir has had a history of thermal stratification and hypolimnetic deoxygena-

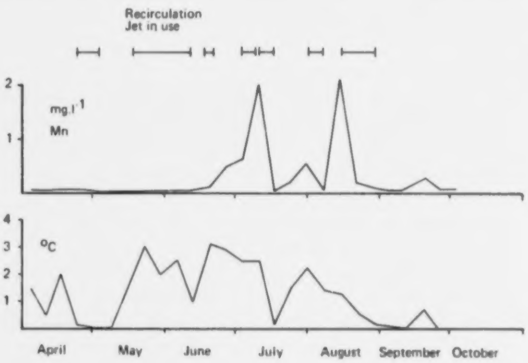

Figure 6. — Dissolved manganese in the bottom water (21 m) and temperature differences between the surface and bottom for a jetted-recirculation system.

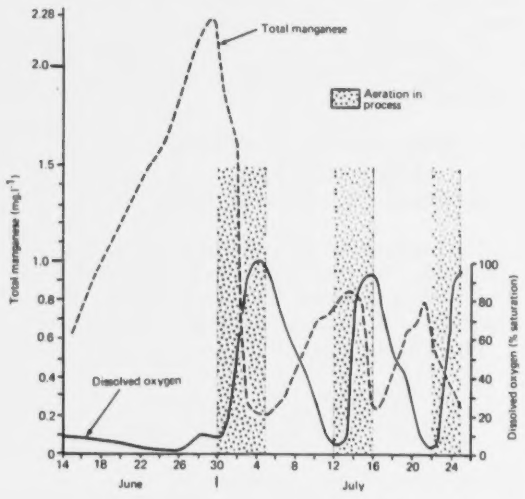

Figure 7. — Dissolved-oxygen and total-manganese levels in the bottom water (21 m) during operation of a perforated-pipe system.

Table 2. — Thermal quarters of water bodies in the southern UK (Steel, 1976).

Quarter	Thermal effect	Months	Water temperature
1	Constant cold	Jan. - March	~ 4°C
2	Constant heating	Apr. - June	~ +5°C month⁻¹
3	Constant warm	July - Sept.	~ 20°C
4	Constant cooling	Oct. - Dec.	~ -5°C month⁻¹

Table 3. — Details of the reservoirs used in the case studies.

Reservoir	Filling method	Mixing system	Volume 10^6m^3	Area ha	Depth max m.	Depth mean m.
1	Pumped storage	Inlet jet	4.5	50.6	11.0	8.9
2	Pumped and Impounding	Recirculation Jet	8.9	115.0	22.9	7.7
3	Impounding	Perforated pipe	1.6	16.6	23.1	9.4

tion giving rise to re-solution of iron and manganese from the bottom sediments. Total manganese concentrations of 3 mg l^{-1} had been common and had on occasions reached 7 mg l^{-1} causing treatment problems at times of high demand. On some occasions the problems were exacerbated by high algal populations in the epilimnion.

A polyethylene pipe, of which an 85 m length was perforated by 0.8 mm dia. holes at 0.3 m centers, was installed in the deepest part of the reservoir near the dam wall. The pipe was attached to a compressor delivering 57mg l^{-1} of free air compressed to 3 bar. The compressor was turned on when the reservoir was already stratified with a column temperature difference of about 7°C. The result of this and the changes in water quality prior to the operation is shown in Figure 6. The total manganese concentration of 2.3 mg l^{-1} was rapidly reduced to 0.2 mg l^{-1} and at the same time the dissolved oxygen content at the bottom of the reservoir reached saturation with a column temperature difference of <1 °C.

When the compressor was turned off, however, thermal stratification re-established and a fall in dissolved oxygen resulted in re-solution of manganese. Two periods of operation were sufficient to improve the water quality. Following this the compressor was used for about 8 hours each day until early autumn to maintain the water quality at a treatable level. The initial high concentration of algae in the surface water was distributed by mixing over the entire depth and this dilution also reduced water treatment problems. Fuel charges for operation of the compressor were approximately £28 per day.

CONCLUDING REMARKS

Inlet jetting and air injection, the latter by means of a perforated pipe system, have been the main methods of artificial mixing in the UK to date. Inlet jetting is ideally suited to pumped-storage schemes but provision for this must be made at the reservoir construction stage. Perforated pipe devices have been used almost exclusively in impounding reservoirs and although they are preferably installed during construction of the dam they may also be installed easily, as and when required, following impoundment.

Experience with both destratification devices has shown that thermal stratification can be prevented from becoming established during the summer and the water quality problems associated with stratification can be controlled.

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CASE STUDIES OF AQUATIC PLANT MANAGEMENT FOR LAKE PRESERVATION AND RESTORATION IN BRITISH COLUMBIA, CANADA

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ABSTRACT

A wide variety of aquatic plant management technologies have been tested and evaluated as part of the British Columbia Aquatic Plant Management Program. These detailed studies are valuable to guide planning and long range management of aquatic macrophytes in diverse aquatic environments. The recent introduction of Eurasian water milfoil (*Myriophyllum spicatum* L.) to British Columbia has caused concern about environmental quality and has interfered with multiple water resource uses in a number of water bodies. Case studies are presented in this paper to illustrate the application of aquatic plant management technologies in preservative or restorative strategies. The technologies are described and in each case study the history and management approaches are summarized. The limitations,costs, and benefits of the strategies have been highly variable but this variability reflects the political, social, and technical complexities of aquatic plant management.

INTRODUCTION

Nuisance aquatic vegetation can threaten or interfere with multiple use of water resources. In British Columbia, there were few reports of problems with aquatic macrophytes until recent years, although some species such as *Elodea canadensis* Rich. in Michx., *Potamogeton crispus* L., and *Ceratophyllum demersum* L. have been recorded as nuisances for some time. However, the introduction and spread of Eurasian water milfoil (*Myriophyllum spicatum* L.) in British Columbia waters during the past decade has created a demonstrable environmental problem which has warranted a substantial management effort. *M. spicatum* has become a widespread management problem in the eastern United States.

While control projects have been necessary in many areas where this plant is now established, the efforts of the B.C. Ministry of Environment are unique because of the diversity, complexity, and comprehensiveness of the management approaches and of the attempts to document the results. A historical perspective of major elements of this aquatic plant management program was presented elsewhere (Newroth, 1979).

The timeliness and suitability of applying technology to specific problems is particularly important in successfully allocating resources for maximum management benefits. Case studies are presented here to exemplify management strategies; technologies and approaches described reflect the changing philosophies and policies developed since 1972.

DESCRIPTION OF TECHNOLOGIES

The B.C. Ministry of Environment has extensively reviewed or field tested a wide range of technologies. Also, the Province has encouraged private enterprise in technological development as well as in original research and development in the following categories:

1. *Prevention* - Since several major regions of the Province are not infested with *M. spicatum* (and experience has demonstrated the difficulty of eradicating established populations), several preventive approaches have been developed. Because boaters are suspected of spreading aquatic weeds, quarantine projects have been used to encourage voluntary public cooperation; boats leaving infested areas are checked (Scales and Bryan, 1979; Dove and Malcolm, 1980). Also, surveillance for *M. spicatum* has been performed at selected noninfested lakes, with emphasis on marina and boat launch areas of those with high public recreational value. Data gathered from the quarantine projects have aided in selecting lakes known to be frequented by boaters who have visited infested water bodies.

2. *Intensive control* - While eradication of Eurasian water milfoil appears to be a remote possibility, the timely application of a variety of technologies may provide a cost-effective means to contain nuisance aquatic plants. Depending on the situation, technologies may be used independently or in an integrated fashion. Detailed descriptions of technologies suitable for intensive control are provided in reports prepared by the Ministry of Environment (Anon., 1978; Bryan, 1980; Maxnuk, 1979; Goddard, 1980). Brief descriptions and major limitations of these techniques are outlined here:

(a) *2,4-D Butoxyethanol Ester*: Laboratory and extensive field testing in the Okanagan Valley lakes since 1976 has indicated the utility and environmental safety of selectively using this herbicide in a granular formulation (Aqua-Kleen 20) (Goddard, 1980). 2,4-D has proven ability to kill most roots and shoots of Eurasian water milfoil in treated areas, and it can be applied rapidly to large areas without inducing plant fragmentation. However, this herbicide cannot be used in localities where the public may become exposed to the herbicide at detectable concentrations following treatment. Also, the most effective applications have coincided with quiescent water conditions. Herbicides can be integrated successfully with other technologies including mechanical harvesting, diver dredging, and rotovating. Although at present the Ministry of Environment only uses 2,4-D for intensive control, in some situations it may be practical for more cosmetic management.

(b) Diver-operated dredges: The use of scuba divers and hand suction dredges has been refined since early trials in 1976. This method is slow, relatively costly, but highly effective in reducing the density of *M. spicatum* infestations (Maxnuk, 1979). The localized activity of divers discourages dissemination of viable plant fragments. The best application of this approach is in small areas, newly populated by Eurasian water milfoil, where visibility is good and non-target vegetation is sparse (ideally in spring and early summer).

3. *Semi-cosmetic or cosmetic control:* In water bodies where *M. spicatum* has become widely distributed or where treated areas are subject to rapid reinfestation from major sources of viable plant fragments, the most practical approach is to minimize the nuisance cosmetically and cheaply. Three types of technology have been developed and used by the Ministry of Environment:

(a) *Shallow water tillage:* Large, shallow, littoral areas may be treated using amphibious vehicles drawing agricultural implements (including plows and discs), to stress and remove plant roots and to lessen the density of nuisance growth. This is most effective when the area is subject to winter drawdown, but obstacles and water pipelines may prevent efficient operations.

(b) *Deep water tillage:* Barge mounted rotovators have been extensively tested in efforts to achieve root removal in water up to 4 m deep (Bryan, 1980). Although this method is relatively slow and costly (as compared with harvesting) and is limited to areas where obstacles or rocks do not interfere with operation, the main nuisance growth may be reduced for several seasons. In certain cases rotovating may be successfully integrated with more long-term management technologies and root removal may be achieved in the winter months.

(c) *Plant harvesting:* Mechanical harvesting provides cosmetic management by removing the tops of nuisance plants. Four large machines are now employed to harvest *M. spicatum* in the Okanagan Valley and a similar machine is used in southern Vancouver Island for control of *Ceratophyllum, Elodea,* and other macrophytes. The main limitations of this technology include the relatively slow speed (as compared with 2,4-D), the need to time harvesting to coincide with nuisance growth, and the repetitive, seasonal nature of the operation. Although future research may demonstrate that frequent or repetitive harvesting may contribute to declines in vigor of *M. spicatum* populations, in some situations the use of harvesters may further spread this plant. Where massive infestations prevail, the harvesting approach is a practical management tool although high capital costs may limit the number of machines.

4. *Passive control — fragment barriers:* A variety of containment devices have been developed for deployment around equipment or to reduce downstream spread of buoyant, viable fragments of *M. spicatum.* Portable floating booms have been used around rotovating and harvesting areas, where warranted, and appear to reduce the escapement of much floating plant material. Stationary fragment barriers have been used in river channels and interconnections between lakes since 1976 and have been developed to the degree that they are generally successful in reducing fragment movement (Stephenson and Baillie, 1980). Practical limitations often restrict the successful application of barriers; they must be maintained routinely, using suction pumps to clear the fragments trapped on the mesh.

CASE STUDIES

As part of the Ministry of Environment aquatic plant management program, over 900 water bodies in British Columbia have been inspected and records made on aquatic macrophytes observed. About 30 lakes with high recreational value are under continuing study. Aquatic plant management technologies have been applied to about 10 additional lakes and case studies of four of these lakes are outlined here to illustrate the approaches, expectations, and results which have been achieved. Figure 1 shows the location of British Columbia lakes now infested with Eurasian water milfoil: Wood, Kalamalka, and Okanagan Lakes are directly interconnected and are located in the Okanagan Valley of south-central British Columbia; Cultus Lake is situated south of the Fraser River in the southwestern part of the Province.

Okanagan Lake

Okanagan Lake is the largest lake in the mainstem lake chain (area 34,800 hectares) and is described as oligotrophic (Stockner and Northcote, 1974). Nuisance growths of *Myriophyllum* in the northern part of this lake were the first reported in British Columbia. The present management program has developed in response to requests for assistance from local agencies. As part of the pilot studies to document aquatic vegetation in Wood, Kalamalka, Okanagan, Skaha, Vaseux, and Osoyoos Lakes (see Figure 1), voucher collections from all these lakes were made in 1972. Although *Myriophyllum* specimens were collected in all these lakes, subsequent analyses of these early collections by chromatography (Ceska, 1977) have verified *Myriophyllum spicatum* only among specimens collected in Okanagan Lake. It is possible that the initial infestation of this plant was in Okanagan Lake and that subsequent downstream spread to other lakes and transport by boaters has led to the infestations recorded later in other areas. Accurate definition of the extent of *M. spicatum* prior to 1972 is impossible, although study of aerial photographs taken prior to the development of the present conspicuous colonies indicated that major expansion occurred in northern Okanagan Lake in the early 1970's.

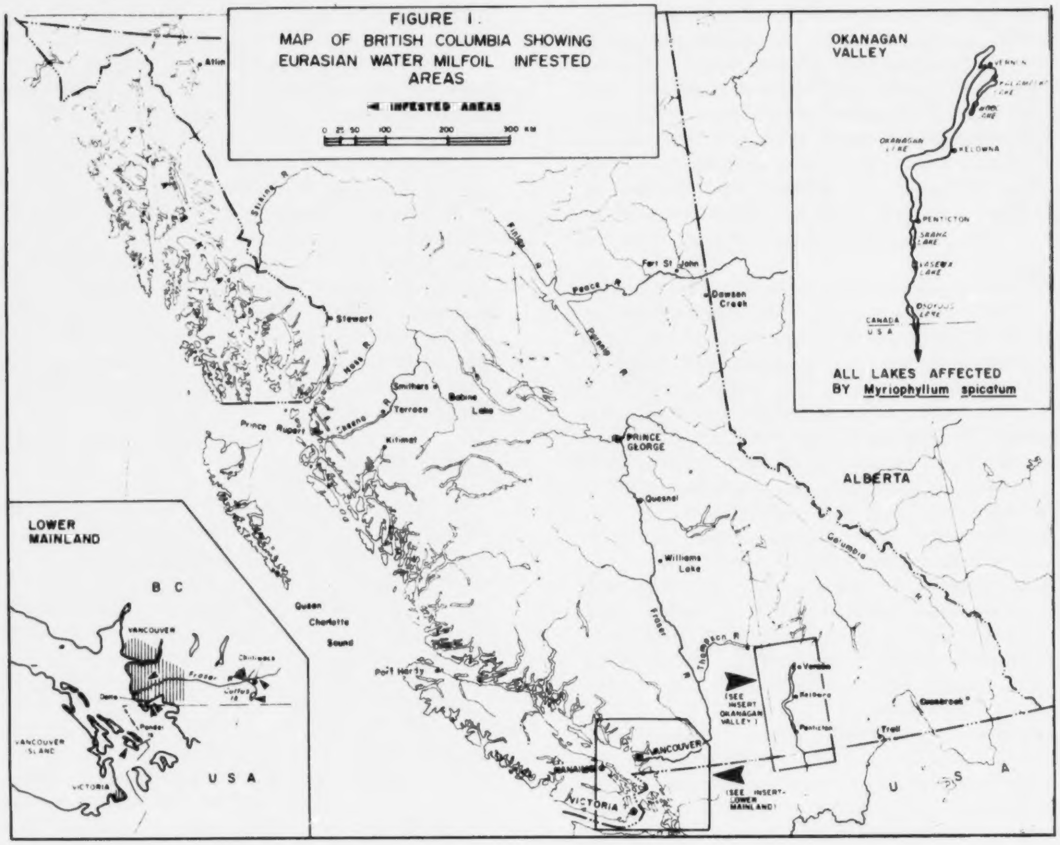
FIGURE 1.
MAP OF BRITISH COLUMBIA SHOWING
EURASIAN WATER MILFOIL INFESTED
AREAS
INFESTED AREAS
0 25 50 100 200 300 KM
OKANAGAN
VALLEY
VERNON
KALAMALKA
LAKE
WOOD
LAKE
OKANAGAN
LAKE
KELOWNA
PENTICTON
SKAHA
LAKE
VASEUX
LAKE
OSOYOOS
LAKE
CANADA
U S A
ALL LAKES AFFECTED
BY Myriophyllum spicatum
Atlin
Stikine R.
Stewart
Nass R.
Finlay R.
Peace R.
Fort St John
Dawson
Creek
Parsnip R.
Smithers
Terrace
Babine
Lake
Prince Rupert
Skeena
Kitimat
PRINCE
GEORGE
Quesnel
ALBERTA
Columbia
Williams
Lake
Fraser
Thompson R.
LOWER
MAINLAND
B C
VANCOUVER
Chilliwack
Fraser
Cultus
L.
Delta
Pender
VANCOUVER
ISLAND
VICTORIA
U S A
Queen
Charlotte
Sound
Port Hardy
Vernon
Kelowna
Penticton
(SEE
INSERT
OKANAGAN
VALLEY)
Cranbrook
Trail
VANCOUVER
NANAIMO
VICTORIA
(SEE INSERT-
LOWER
MAINLAND)
U S A

Figure 1. — Map of British Columbia showing Eurasian water milfoil-infested areas.

Although localized surveys of aquatic plant populations were performed in 1972-74, a more comprehensive survey was performed in 1975 (Nijman, 1976). This documentation has been expanded each year since 1975; Table 1 indicates the changes in area occupied by *M. spicatum* in the lakes discussed here.

In retrospect, it appears that early containment and immediate intensive control efforts should have been considered seriously for Okanagan Lake. However, limited resources were available in 1972 and the major nuisances caused by the rapid spread and growth of *M. spicatum* were not publicly recognized until 1973 and 1974; in many ways, the nuisance potential of *M. spicatum* was underestimated. Despite the experiences of agencies in other areas (especially Florida and the Tennessee Valley) with this plant, funding necessary to begin effective work on the growing problem was not forthcoming for several years.

Recent applications of aquatic plant management technologies in Okanagan Lake are summarized in Table 2. Because the potential negative impact of Eurasian water milfoil was clearly apparent in 1974, resources were made available in 1975 and 1976 for extensive testing and monitoring of:

1. Contact herbicides, diquat and paraquat in 1974 and 1975 (Bryan, et al. 1977) and the systemic herbicide 2,4-D (Lim and Lozoway, 1977).
2. Bottom barriers (Armour, et al. 1979).
3. Mud Cat hydraulic dredge (Bryan, 1978).
4. Three types of rotovating machines and a hydraulic washing device (Maxnuk, 1979; Bryan, 1980).

Most of these treatments were performed in popular recreational areas and somewhat relieved nuisance conditions, but as shown in Table 1, this activity coincided with a rapid expansion (1975-1977) of Eurasian water milfoil in Okanagan Lake. Because of uncertainty, lack of experience as to the most appropriate technology, and lack of funding, no

Table 1. — Recent records of *M. spicatum* in case study lakes.

Lake	Approximate Littoral Area (ha)	Area affected by *M. spicatum* (ha)				
		1975	1976	1977	1978	1979
Wood	85	<1	<1	3	12	17
Kalamalka	145	<	3	11	12	4
Okanagan	1,930	233	288	358	398	403
Cultus	37	?	?	13	17	16

concerted effort was planned until later. Thorough evaluations of the trials showed some benefits from the treatments but reinfestation from outside areas and regrowth of *M. spicatum* roots remaining in treated areas indicated that the large scale use of 2,4-D was the only practical technology that might substantially reduce Eurasian water milfoil in Okanagan Lake. Also, containment and overall control in Okanagan Lake became a lower priority because of the rapid growth of *M. spicatum* in Kalamalka and Wood Lakes during 1975-1977.

In 1978 nearly 400 hectares of shoreline in Okanagan Lake were colonized by Eurasian water milfoil and many public and private recreational areas were affected to some degree. A major control program was planned for 1978 and designed to achieve a major reduction of the primary nuisance colonies in Okanagan Lake; however, the Herbicide Use Permits necessary for large scale use of 2,4-D were not received. As shown in Table 3, major efforts were made using harvesters and other machines but only about 12 hectares of experimental 2,4-D treatments could be performed. Reluctantly, it was recognized in 1978 that no practical options for overall control of Eurasian water milfoil in Okanagan Lake were available. The 1979 and 1980 programs have been designed to relieve nuisances at relatively low costs using harvesters only. Basic harvesting costs (excluding administrative and capital costs) averaged between $1,500 and $1,900 per hecatare in 1978 and about $1,200 per hectare in 1979. Because efforts have continued to improve these machines (reduce downtime and increase overall harvesting efficiency) it is hoped that the 1980 costs will be slightly lower than those documented in 1979. Because of the ongoing need for harvesting, local authorities have been encouraged to share costs of cosmetic harvester operation, with 75 percent of the operating costs borne by the Province.

Kalamalka and Wood Lakes

The Kalamalka-Wood Lakes Basin in the Okanagan region has been the subject of intensive management study; both lakes (see Figure 1) are considered of high value for recreation and water supply purposes (Anon. 1974). Both lakes support populations of Eurasian water milfoil although they are completely different in character (Wood Lake, area 930 hectares, eutrophic: Kalamalka Lake, area 2,590 hectares, oligotrophic) (Stockner and Northcote, 1974). Preserving the aesthetic beauty of Kalamalka Lake is considered a very high priority by government and residents of the area. Because dense aquatic plant growth would impair the attractive white marl littoral areas of Kalamalka Lake and interfere with beach use and boating, aquatic macrophyte control has been recognized as an important objective (Anon. 1974). Wood Lake discharges through a short canal to Kalamalka Lake downstream so successful management requires simultaneous attention to both lakes.

Independent of the aquatic plant management program, water quality studies to preserve Kalamalka Lake and improve Wood Lake have been sponsored by the Province and Federal agencies. These studies have revealed a trend of increasing water transparency over the past 3 or 4 years (Nordin, 1980). This phenomenon may be linked to the dramatic increase of area affected by *M. spicatum* in Wood Lake between 1976 and 1979 (see Table 1).

Table 2. — Summary of aquatic plant management technologies applied to Okanagan Lake.

	Year of application and approximate area treated (ha)				
Technology	1975	1976	1977	1978	1979
Harvesting	15	Nil	45	55	47
Rotovating	Nil	55[1]	4[2]	4	Nil
Dredges	2.9	Nil	Nil	Nil	Nil
a) Mud Cat	2.9	Nil	Nil	Nil	Nil
b) Diver-operated	Nil	Nil	Nil	1.0	Nil
Herbicides					
a) Paraquat/Diquat	1.6	Nil	Nil	Nil	Nil
b) 2,4-D (B.E.E.)	Nil	1.2	7.0	12.0	Nil

[1] Area treated by tractor drawn rotovator (5 ha), amphibious rotovator (24 ha) and barge mounted rotovator (26 ha).

[2] Barge mounted rotovator only from 1977 on.

Table 3. — Summary of aquatic plant management technologies applied to Kalamalka and Wood Lakes.

		Year of application and approximate area treated (ha)			
Lake	Technology	1976	1977	1978	1979
Wood	Diver-operated Dredge	Nil	Nil	11	5
	2,4-D (B.E.E.)	Nil	Nil	Nil	7
Kalamalka	Rotovating	0.5	10	10	Nil
	Diver-operated Dredge	Nil	5	24	28
	2,4-D (B.E.E.)	Nil	Nil	<1.0	13

Although minor nuisances caused by growth of *Potamogeton crispus* had been reported as early as 1972, surveys for aquatic plants did not locate *M. spicatum* until 1974. The first collections (1974) of *M. spicatum* in Kalamalka Lake were confirmed several years later by chromatography (Ceska, 1977). Because of the close morphological similarity of this species to *M. exalbescens* Fern., positive identification was first confirmed in 1975 when small populations of characteristic vigorous growth were located at the north end of Kalamalka Lake and the south end of Wood Lake (Nijman, 1976). Both populations had developed adjacent to boat launching and marina facilities.

Prior to the observation of established colonies of Eurasian water milfoil in these lakes, posting of signs to discourage the introduction of fragments and immediate efforts to eradicate *M. spicatum* were recommended (Newroth, 1975). These recommendations coincided with extensive technological testing which began in 1975; pilot scale rotovating was performed in the fall, 1976, in the most dense *M. spicatum* colony in Kalamalka Lake. Table 3 summarizes the applications of management technologies in Wood and Kalamalka Lakes. As shown in Table 1, major expansion of Eurasian water milfoil was documented in Kalamalka Lake between 1975 and 1976, but no major change in *M. spicatum* growth was seen in Wood Lake during this interval.

Details of the management efforts in Kalamalka Lake are presented elsewhere (Bryan, 1976; Maxnuk, 1979; Armour, et al. 1980): the objectives, efforts and results achieved in this lake are illustrative and are reviewed briefly here. Following the trials with a prototype rotovator in Okanagan Lake in 1976, about 4 hectares of moderately infested area of Kalamalka Lake were treated in early 1977. A new, improved rotovator treated about 6 hectares of Kalamalka Lake in the fall, 1977. Also, several versions of diver-operated dredges were evaluated and used to treat about 5 hectares of littoral area in this lake during the spring and fall, 1977. These efforts and subsequent treatments of large areas of Kalamalka Lake by dredging (24 hectares) and rotovating (10 hectares) in 1978 were intended to achieve the maximum possible degree of control of Eurasian water milfoil. However, reinfestation of previously rotovated areas was observed despite short term effectiveness of up to 90 percent root removal. The diver dredges concentrated in 1977 on removing roots and shoots missed by rotovating, and achieved up to 97 percent effectiveness (Maxnuk, 1979). In 1978, larger diver dredges were made available and continued to minimize regrowth in previously rotovated areas of Kalamalka Lake. Also, these improved machines with larger crews were deployed widely to search for and remove small, new Eurasian water milfoil infestations on the littoral zones of Kalamalka and Wood Lakes.

In the case of Kalamalka Lake, the combination of early and extensive integrated control efforts in 1977 and 1978, combined with good underwater visibility, appeared to have stabilized Eurasian water milfoil populations (see Table 1). However, it was recognized that although rotovating and diver dredging could give very good cosmetic control and prevent nuisance conditions, fragmentation of *M. spicatum* and the slow speed of mechanical treatments would lead to further spread and were unlikely to achieve the needed reduction in size of dense major colonies. In 1979, 13 hectares of Kalamalka Lake were treated with 2,4-D; most of this target area was the same shoreline that had been treated previously with machines. Diver dredging in 1979 (28 hectares) was concentrated on eliminating new colonies in Kalamalka Lake.

Operating costs for rotovating have varied widely, depending on the stage of mechanical development and suitability of the area treated. Excluding administrative, overhead capital costs, and monitoring expenses, the last major rotovating work (Kalamalka Lake, 1978) averaged $2,200/ha. In 1979, the diver-operated dredging in Wood and Kalamalka Lakes was estimated to cost about $1,600/ha. Herbicide treatments have been relatively expensive because of high administrative costs associated with the research and monitoring objectives of all treatments to date. Excluding some of these costs, but including costs of the monitoring and alternate water supply systems required by regulation of the Herbicide Use Permit, the 2,4-D treatments averaged $4,300/ha for Kalamalka and Wood Lakes in 1979.

Documentation of aquatic plant populations in Wood Lake has been frustrated by poor visibility until recent years. It is suspected that Eurasian water milfoil growth and expansion were limited by turbidity until about 1977. At this time, a major increase in Eurasian water milfoil was documented and in 1978 diver-operated dredges were deployed to attempt control of these populations. Unfortunately, a fourfold increase in *M. spicatum* was recorded in 1978 (see Table 1) and plans were made for major herbicide applications in 1979. Approximately 7 hectares of affected area at the south end of Wood Lake were treated with 2,4-D in 1979 and limited diver dredging was performed in an additional 5-hectare area to reduce further fragmentation and expansion. In 1979 and 1980, aquatic plant fragment barriers were installed in the canal between Wood and Kalamalka Lakes and maintained to minimize fragment movement with the current from Wood to Kalamalka Lake.

At this time, the Ministry of Environment is continuing intensive management of both Kalamalka and Wood Lakes. Because the apparent success in achieving a major reduction in 1979 of area affected by *M. spicatum* in Kalamalka Lake has been maintained in 1980, spot treatments using 2,4-D and intensive diver dredging are continuing. It is hoped that containment of Eurasian water milfoil can be maintained and that overall expenses of this preventive maintenance can be reduced. Monitoring of the Eurasian water milfoil colonies must be continued unabated to ensure that adequate resources are allocated to preservation. The dramatic expansion of Eurasian water milfoil recorded in Wood Lake in 1979 has continued in 1980 and major 2,4-D treatments of substantial shoreline areas are being contemplated. Because of water use constraints and the large colonies of plants, it is feared that extensive management of Wood Lake may be more difficult than Kalamalka Lake.

Cultus Lake

Cultus Lake is an oligotrophic lake of considerable recreational importance because of its proximity to the City of Vancouver and other major urban areas in the Lower Mainland region of southeastern British Columbia. The littoral area is relatively small (about 37 hectares) compared to the surface area (630 hectares). Aquatic plant surveys were first performed in this lake by the Ministry of Environment in 1977 as part of baseline studies throughout southern British Columbia. Surveys in 1977 and 1978 showed that a number of other water bodies in the Lower Mainland were affected. Because there were no documentation studies prior to 1977, there are no adequate records to indicate the probable sites of initial infestations in the Lower Mainland. Approximately 13 hectares of littoral areas of Cultus Lake were found to be affected by *M. spicatum*. The populations located in Cultus Lake in the fall of 1977 appeared to have resulted from recent introduction and most plants were distributed then in the marina area and in a downstream direction toward the lake outlet.

Since numerous sites in Cultus Lake and several adjacent water bodies were already populated with *M. spicatum*, major control efforts directed toward containment or eradication did not appear practical at the outset. Because of public concern from residents and local government and recreational agencies associated with Cultus Lake, various options for managing the existing populations in this lake were reviewed. Since *M. spicatum* had become established in the river outlet of Cultus Lake, fragment barriers were considered of limited value. Control by harvesters, rotovators, or other tillage equipment was considered. However, the relatively sparse infestations were not suitable for harvesting, and further spread of fragments was feared; also, *M. spicatum* colonies were established in many areas where obstacles (such as docks and submerged logs) or rocky bottom conditions would preclude effective treatment with large machines. The use of 2,4-D herbicide and/or diver-operated dredges appeared to be most practical. Because of controversy about use of 2,4-D and concern about the public use of water from the lake, diver dredging was selected. In cooperation with a local group, the Cultus Lake Milfoil Action Committee, formed in 1978, the Ministry of Environment performed feasibility studies in 1978 to assess the use of a diver-operated dredge.

The objective of the 1978 trials and subsequent treatments performed under a cost-sharing agreement (75 percent Provincial: 25 percent local funds), has been to reduce the density of *M. spicatum*; this may have reduced the rate of spread. In 1979 about 1.3 hectares were treated at a cost of about $50,000 and in 1980 about 2 hectares are proposed for treatment at a cost of approximately $60,000. Because of uncertainty about the future impacts of *M. spicatum's* continued expansion, and to provide detailed information on the treatment method and its effectiveness, the Cultus Lake operational program has been monitored closely.

The Cultus Lake example is intermediate between the Kalamalka-Wood Lakes case (intensive management) and the Okanagan Lake example where only cosmetic control appears practical. Local circumstances, including factors such as recreational demand, local cooperative interest, and the present and potential impacts of plant growth, have determined the management approach. As long as the Eurasian water milfoil colonies in Cultus Lake appear to be relatively stable in extent and density, semi-cosmetic management at minimum cost appears to be justified. Because there remains concern that *M. spicatum* may be transported to other, noninfested lakes by the heavy traffic of boaters who utilize Cultus Lake, efforts to clear plants adjacent to the boat launch ramps are a high priority. Public information and a voluntary aquatic plant quarantine check station will be employed by the Ministry of Environment in 1980 to reduce the chance of further spread by boaters.

DISCUSSION

The aquatic plant managment experiences and objectives of the B.C. Ministry of Environment are illustrated by the case studies presented here. Objectives of the comprehensive program that has developed in British Columbia include:

1. Identification and evaluation of the conflicts to multiple use of water bodies caused by unwanted growth of aquatic macrophytes (and especially exotic species such as Eurasian water milfoil).
2. Research and evaluation of all practical aquatic plant management technologies and application strategies.
3. Response to public need and relief of nuisance conditions in an environmentally acceptable and effective manner, with documentation of the results.
4. Prevention of undue further spread of Eurasian water milfoil to presently unaffected water bodies in British Columbia.

Experience has shown the complexity and high degree of difficulty of successfully achieving these objectives. Some of the major considerations and constraints are summarized in context with the case studies.

No management can be effective unless the problem is clearly identified and public and political sentiment support both the need and the means. Because of general ignorance about the identification, biological capacity, and ecological impacts of introduced aquatic plants, there has been uncertainty about the nature of aquatic plant problems. The initial reports of Eurasian water milfoil in Okanagan Lake were followed by a period during which the plants were identified correctly and local nuisance conditions were experienced and documented (1972 to 1975). The next phase (1975 to 1977) included the testing and development of technologies believed most suitable to control the infestations. Another very important preventive phase, a major effort to contain the relatively small initial infestations, was not initiated in time in Okanagan Lake. Consequently, it now appears that the initial infestations in Okanagan Lake were rapidly spread downstream by water movement and possibly to Kalamalka, Wood, and Cultus Lakes (and others) by boaters. Spread of potential nuisance organisms is difficult to monitor, especially in the aquatic environment, and infestations of aquatic weeds are virtually impossible to locate at an early stage.

Experiences with aquatic plant management technologies have demonstrated that complete elimination of exotic aquatic plants is exceedingly difficult. Also, the degree of difficulty increases in proportion to the area affected and the number of sources of reinfestation. In addition to the costs of technological development, surveys to locate new infestations and documentation of the trials and continuing evaluations of ongoing work (i.e., harvesting benefits in Okanagan Lake; measurement of diver dredging effectiveness in Cultus Lake) are also costly.

To respond to predictable and projected management needs, considerable staff with diverse skills must be assembled and trained. This organization must implement consistent, long-term policies in consultation with local authorities to secure public support and cooperation and to share management responsibilities and expenses. The Cultus Lake project involves local agencies in selection and cost-share funding of appropriate management techniques. Responsibilities for cosmetic control also will be turned over to local authorities in the Okanagan Valley. The Province will provide technical guidance and secure major funding for the necessary implementation work.

Needs for long-term aquatic plant management, and a preventive approach wherever practical, are apparent because of the importance of water-based recreation to the general public and to the tourism industry.

However, data are not available on the dollar value of certain beach or marina areas. These data will be important to determine and assess priorities for future management. Comparisons of cost and benefit of various technologies is difficult and may not be particularly worthwhile until routine procedures are established. Each technology is a somewhat specific tool appropriat to certain locations, situations, and seasons; combinations of technologies gained from the intensive management of Kalamalka and Wood Lakes may be applicable to some other presently infested lakes or to noninfested water bodies. These lakes present the best opportunity to achieve a high standard of control at a minimal cost of annual maintenance.

The case studies illustrate the interactions of planning, field observation, and deployment of appropriate technologies. Effective management requires a successful combination of comprehensive policies and long-range planning with technical knowledge, adequate funding, and public support. Operational programs must be developed within a framework of clear and realistic objectives. Continual feedback on the needs and benefits of the restorative or preservative program is essential to guide policymaking and implementation. All of the cases described here were reactive situations. Experience has shown that the rapid spread and potential nuisance impacts of exotic aquatic plants should not be underestimated. Also, control and management, once an exotic plant is established, may be very difficult and costly, or unsatisfactorily cosmetic and require an ongoing commitment. This reality may pose a heavy burden for senior government and the local authority or taxpayer.

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GERMAN EXPERIENCE IN RESERVOIR MANAGEMENT AND CONTROL

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ABSTRACT

In Germany reservoir management and control begins prior to construction. Monitoring programs are set up for chemical and biological investigations of the tributaries. From the results various control measures are derived, such as diverting of local wastewater inputs, designing a treatment plant, etc. A traditional control measure has been to build pre-reservoirs. Their original purpose was silt sedimentation, but more recently it has become obvious that pre-reservoirs act as nutrient sinks and therefore can protect reservoirs from eutrophication. Another control measure is establishing protective zones with legal restrictions for various activities. In cooperation with forest experts, rules for afforestation of the slopes around reservoirs have been worked out. Different control measures are applied to protect reservoirs from eutrophication. In the case of point sources of nutrients wastewater is pumped through a pipeline to a treatment plant which is situated downstream from the reservoir. In the case of diffuse sources nutrients are removed from the tributaries. In small tributaries, this is done by constructing seepage trenches or filtering through aluminum oxide columns; chemical precipitation and subsequent filtration in a plant or sedimentation in the reservoir itself are used in large tributaries. Only hypolimnetic aeration is widely used as a control measure within the reservoir.

INTRODUCTION

There are two aspects of reservoir management. Either management relates to water quantity or it relates to water quality. Although management of water quantity is a well-established practice, methods for water quality management are still being developed. Water quality problems are of special importance in the drinking water reservoirs. These are reservoirs from which water is pumped directly to a treatment plant which supplies the public with potable water. This paper deals exclusively with management and control of water quality in drinking water reservoirs, above all, considering the problem of eutrophication.

THE IMPORTANCE OF WATER QUALITY PREDICTIONS

Management and control of a reservoir should begin prior to its construction. This means that as many measures as possible should be applied to protect water quality when the reservoir is still being planned. In Germany experience has shown that control measures are much easier and less costly to put through during this early stage than later.

The most important tool for such management is the prediction of water quality of the reservoir which is being planned. Up to rather recent times such estimations were based on a saprobiological survey of the main tributary and an analysis of only a few randomly taken water samples. It was assumed that these few samples were representative for a long period of time and that the water quality of the reservoir should not differ considerably from the quality of its tributaries. This concept failed when reservoirs could no longer be created in remote woodland; for example, Wahnbach Reservoir had to be built in densely populated agricultural land (Clasen, 1979).

Today, predictions of reservoir water quality are based on detailed monitoring programs. The control of point sources of nutrients is based on the results of such investigations. For concentrations of nutrients which cannot be controlled in this way the trophic status of the planned reservoir is estimated using Vollenweider's approach (1976). This prediction is then used to design the planned treatment plant. The concept which is described here is at present applied to several reservoirs which are just being planned or built in FRG.

REMOVAL OF SOIL AND TREE TRUNKS

Water quality of a reservoir not only depends on water quality of the tributaries but also on the quality of the basin. This refers mainly to the soil,which may release nutrients and undesirable organic substances. As a consequence, removing the soil and tree trunks may be desirable, but financially prohibitive. Since the impact of soil is strongest during the first year after impounding in such cases water which had been dammed up might have to be discarded in the course of one year. Very recently this happened in a new reservoir in the Federal Republic of Germany.

INTAKE DEVICES OF TREATMENT PLANTS

Experience has shown that treatment plants should be able to take raw water from several different levels. It is therefore possible to avoid concentration peaks caused by fast flowing floods, development of algae, or release of unpleasant substances from the sediment. Of course intake at different levels is advantageous only if at least a minimum of monitoring of water quality in the reservoir takes place. Recently the author of this paper was consulted by a treatment plant which had run into great trouble because extraordinary numbers of diatoms penetrated the filters. The reservoir had not been monitored before and there was an intake device at only one level. An *in situ* turbidity profile revealed that this coincided exactly with the peak of the heavy diatom bloom. In this case management had to consist of waiting until the breakdown of the bloom.

Different intake levels give several advantages. During stratification periods raw water can be taken close to the bottom in order to renew the water overlaying the sediment. As soon as too much manganese, etc., is released from the sediment one has to switch to the next higher intake level. In several drinking water reservoirs in the FRG experience has shown that the hypolimnion should not be too small as compared to the amount of raw water which is processed during the period of stagnation (Bernhardt, Clasen, and Nusch, 1973). Otherwise, the zone of decay (hypolimnion) will become too small toward the end of the stratification period as compared with the production zone (epilimnion), which means an overloading of the mineralization capacity. These observations show that sometimes quality problems are closely related to water quantity. This aspect should be considered in a very early stage of planning.

PRE-RESERVOIRS

Traditionally, German reservoirs are provided with pre-reservoirs in which at least the major tributaries are impounded. The original purpose of these pre-reservoirs was to retain silt. Recently, however, it has become evident that they also retain nutrients and thus to some extent protect reservoirs from eutrophication. Basic work on this has been done mainly by Bendorf, Putz, and Henke (1975) in the German Democratic Republic. As opposed to Vollenweider, Bendorf and co-workers only consideredo-PO_4. Whereas Vollenweider mainly considered water bodies with retention times of more than 1 year, Bendorf deals with water bodies with retention times of less than 3 months.

The phosphorus elimination in pre-reservoirs is based on an extensive amount of bioproductivity. Phosphorus becomes fixed in the biomass in the pre-reservoir. This biomass is retained to a great extent by sedimentation. The phosphorus remains fixed on the bottom if there is sufficient oxygen available for this purpose.

The P-elimination rate can be predicted if the following parameters are known:

a) o-PO^4 ortho-phosphate concentration in the tributary P(μg/l P)

b) Water temperature T (°C) in the reactor (pre-reservoir) V

c) Average light intensity in the uppermost 3 m layer of the pre-reservoir 1 m(cal/cm^2.d)l

d) Calculated water retention time t(d)

e) Actual water troughput q(m^3/d)

From the parameters listed under a - d the critical retention time t can be calculated. t is identical with t , if the output loss of phytoplankton is not greater than the production rate.

$$\frac{1}{\left(\frac{P}{0.5+P}\right)\left(\frac{I}{10+I}\right)\left(\frac{T}{20}\right)} = t_{krit}$$

From the ratio of actual (t) to critical retention time the elimination rate of o-PO :

$$\frac{t_c}{t_{krit}} = \frac{V_R}{q \cdot t_{krit}} = n$$

Since Bendorf's formula also includes light-intensity and temperature, estimations can be made for every season. In winter when there is little bioactivity the elimination figure for O-PO_4 is much smaller than during summer. Wilhelmus, Bernhardt, and Neumann (1978), who carried out similar investigations at various pre-reservoirs in the FRG, calculated for Wahnbach pre-reservoir an average o-PO_4 elimination capacity of 40 percent over a total examination period of approximately 2 years. For short periods, when the inflow was very low this even increased to 90 percent.

Nusch and Koppe (1975) state that the P-elimination rate of the pre-reservoir of the Moehne-Reservoir is 40 to 80 percent. The elimination rate of another pre-reservoir which did not have such a heavy P-load was 8 to 65 percent, depending on the time of year. These examinations show that the retention time of the water in the pre-reservoir must be at least 15 days during normal water flow to achieve a 60 percent P-elimination. For calculations only the upper 3 to 5 meters of water are taken into account. The fact that elimination capacity is low during the winter is a disadvantage of the biological phosphorus elimination process.

SEEPAGE TRENCHES

Although the beneficial effect of pre-reservoirs has been known for decades, the mechanisms involved have been understood only recently. This has resulted from another method of reservoir control, the seepage trenches. These are a means of reducing the P-content of small streams rich in nutrients which originate chiefly from diffuse sources. Phosphorus is eliminated when the water passes through the ground and this process is even more effective in predominantly fine-grained sandy clay. The phosphorus becomes fixed in the upper soil layers.

This method has been successfully used for years by the Wuppertal Municipal Works for protecting the Kerspe Reservoir, a drinking water reservoir in the Bergisches Land. This reservoir contains 15.5 million m³ and has eight important tributaries draining a catchment area with an average of 60 percent woodland. The rest of the area is made up of fields and pastures. The population numbers 623 who live in scattered settlements. The eight tributaries flow via a pre-basin into seepage trenches. The o-phosphate ion content of these streams is between 3 and 60 µg/l, 17 µg/l on an average. The total phosphorus concentration amounts to an average of 40 to 60 µg/lp.

According to Grau's examinations (1978), the use of seepage trenches and pre-basins decreases the dissolved o-phosphate ions from an average 17 µg/l to an average 7 µg/l which corresponds to 61 percent elimination. The seepage capacity of the one stream is 55 m³/hr and that of the other is 250 m³/hr at a maximum, thereby guaranteeing complete treatment of the influent, at least during the summer. In another seepage trench, the average o-phosphate ion concentration is reduced by 50 percent from 25 µg/l P to 13 µg/l P. During the total period it was reduced by 37 percent from an average of 19 µg/l P to 12 µg/l P. This shows this method can reduce the o-phosphate ion concentration in the influent to a figure which can hardly cause rapid eutrophication.

Numerous examinations have show that PO_4^{-3} ions are almost completely chemically fixed by filtration owing to the loam and clay in the upper soil layers. It is a known fact that the cleansing process of ground filtration depends on the following factors:

1. The chemical process of sifting.
2. The physical chemical process of adsorption and ion exchange.
3. The biological process of enzyme reactions.

The range of the processes in these seepage trenches is an extremely wide one. Particular mention should be given to the adsorption mechanism. Those materials can be considered adsorbents which have large interior and exterior surfaces, especially clay and bleaching ores. The kinetics of all these adsorption processes is based on attaining an equilibrium.

Cation and anion exchange are of particular importance for phosphate elimination. Anion exchange is of special importance and it is determined by the anion exchange capacity of the soil. Phosphate ion exchange depends first on the number of hydroxyl groups in the clay and on the atoms in the octahedron centers. Aluminum or iron atoms are particularly good for this purpose but quartz (sand) does not bind phosphate ions and has only low anion exchange capacity. When there is a good deal of humus in the soil, its capacity to bind phosphate decreases because the organic substance has a higher cation exchange capacity than clay minerals.

The anion exchange of a type of soil may be overshadowed by precipitation reactions so that the actual processes can be separated from each other only with difficulty. For this reason one often speaks of anion sorption. Unlike cation exchange, anion sorption increases when the pH goes down. According to Schwertmann's and Knittel's examinations (1972), at a pH of 4.6 it is 28 percent larger than at a pH of 5.5. Biological processes are not of considerable importance here. However, there is a problem with macrophyte growth clogging the walls of the trenches.

Grau's examinations explained the requirements for arranging seepage trenches. To eliminate phosphorus, nitrogen, and trace metal compounds, the ground's clay content is important but it also has a detrimental effect on the infiltration capacity. One has to achieve an optimum state between the two. A sloping hilly site with parallel underground strata is recommended. The required length of the seepage trench is established from the seepage quantity, the permeability of the filter material, and the selected width of the trench. It should not be more than 2 to 3 meters wide so as to be able to loosen or clear off the upper filtration layer. With a trench length of 700 meters and a filtration speed of 1 to 2 meters per day, the maximum seepage capacity is 2,400 to 4,000 m³ per day. It is also advisable to have a pre-basin so as to be able to control maximum seepage. Maintenance costs for this type of system entail cleaning the pre-basin about every 5 years and other service expenses. This is an economic, natural, and satisfactory way of eliminating phosphorus.

ALUMINUM OXIDE COLUMNS

A new method of removing phosphorus from small tributaries is the application of aluminum oxide columns. This is possible mainly in small tributaries rich in phosphorus where the amount of run off does not fluctuate too much (e.g., fish-breeding basins, fed by spring deliveries in the upper part of streams which are more or less of a constant size). At the moment, tests are being carried out on the Wahnbach Reservoir to reduce phosphorus in the effluent of a trout-breeding basin by filtering it through an aluminum oxide column. It is to be filtered to the extent that it will no longer have a detrimental effect on the reservoir water. This very simple plant which has been working under field conditions is built so that it cannot freeze up even at temperatures of -20°C nor can it become blocked with leaves or other detritus. It has been in operation for several months and has reduced the P-content from an average of 370 mg/m³ to an average of 20 mg/m³.

PHOSPHORUS ELIMINATION FROM TRIBUTARIES USING CHEMICAL METHODS

In the case of great tributaries seepage trenches or aluminum oxide columns cannot be applied. Phosphorus must then be removed by flocculation and filtration. A comprehensive way of preventing progressive eutrophication in water bodies is by eliminating phosphorus from the reservoir's main tributary. This method can be applied where the lake or reservoir only has one or at the most two main tributaries. It is used on the Wahnbach Reservoir (Berhnardt and Schell, 1979; Hotter, 1979). Intensive examination of data from the Wahnbach Reservoir has shown that 80 to 90 percent of the total phosphorus which flows into the reservoir is carried in by the river Wahnbach, the reservoir's main tributary.

To change the reservoir from eutrophic to an oligotrophic or mesotrophic state, the phosphorus entering the reservoir had to be reduced by approximately 90 percent. The average total phosphorus concentration of the river Wahnbach is reduced from 90 μg/l to 5 μg/l P. The chemical process used entails precipitation with iron-III-salts, subsequent flocculation, and multi-layer filtration. The plant has a maximum capacity of 18,000 m^3/hr and a filter surface of 1,100 m^2. A new method of precipitation, flocculation, and filtration was developed, the 'Wahnbach System'. This system not only eliminated the phosphorus from the water but an average of 77 percent of the COD, more than 50 percent of the dissolved organic compounds as well as 99 percent of the bacteria. This system therefore gives an additional improvement to the quality of the water in the Wahnbach Reservoir.

The method of adding iron- or aluminum salts directly to a lake or its tributaries so that the phosphorus compounds can precipitate is widely used in Dutch reservoirs but in Germany is applied only to a part of the reservoir 'Halterner Stausee' (Gelsenwasser AG).

TERTIARY TREATMENT

In the case of point sources the phosphorus elimination processes carried out usually are connected with normal mechanical-biological sewage treatment. In the FRG this method is applied in Moehne Reservoir where the outflow of the sewage treatment plant is diverted to the pre-reservoir.

DIVERSION CHANNELS

The use of diversion channels (circular channels) is a widespread means of preventing wastewater from entering lakes and causing eutrophication. In the FRG this method is applied mainly to natural lakes but also to at least two reservoirs: Soese Reservoir and Innerste Reservoir, both situated in the Harz mountains in Northern Germany (Harzwasserwerke des Landes Niedersachsen).

AERATION

Aeration is a method applied to various lakes but especially reservoirs (Wahnbach Reservoir and Ennepe Reservoir in the Federal Republic of Germany). The main aim of aeration is to artificially restore the equilibrium between bioproduction and respiration capacity. Special apparatus is used to pump oxygen (atmospheric air) into the tropholytic zone and particularly into the sediment-water zone to compensate the oxygen depletion processes there and to maintain aerobic conditions in the micro-layer during summer stagnation.

Thus aeration represents a process which prevents phosphorus from being transported from the sediment into the free water zone and thus into the production zone. By creating oxidative conditions in the sediment-water using artificial aeration phosphorus can be fixed in the sediment to which it has been carried along various paths and the sediment actually becomes a phosphorus trap. Artificial oxygen input into the sediment also means that the organic algal substances collecting there can mineralize. This prevents oxygen-depleting substances from accumulating.

The extent of the reductive solution of manganese and iron compounds, the conversion of nitrate to nitrogen or to ammonium ions, and the conversion of sufate ions to hydrogen sulfate all depend on the size of the reduction potential. If these reduction processes cannot take place then concentrations of dual manganese and iron ions cannot increase and ammonium ions cannot form in the water in the tropholytic zone. There can be no formation of sulfide ions and methane gas cannot exist. These interconnected processes have a considerable influence on the size of the phosphorus cycle in a lake and thus on plankton production (Ohle, 1953).

From experience gathered during work on the Wahnbach Reservoir aeration must be carried out to such an extent that the oxygen content of the micro-layer is not allowed to sink to below 3 mg/l oxygen during stagnation. If this is achieved, then one can be sure that the upper layer of the sediment which is in contact with the water body has sufficient oxidation potential. Reduced ions and phosphate ions will be prevented from being released into the free water zone. The process of hypolimnic aeration which was developed by the Wahnbach Reservoir Association over 10 years ago has proved particularly effective (Bernhardt, 1978).

OTHER CONTROL METHODS WITHIN RESERVOIRS

Facilities for other control measures within the reservoirs are rather limited. The application of herbicides such as copper sulfate to control algae has never been considered in Germany. Growth limitation by artificial mixing which is widely used in the Netherlands and the United Kingdom is not applied in the FRG for several reasons. Since the non-biotic extinction coefficient is usually much lower than in Dutch or English reservoirs the growth-limitation by artificial mixing would be also much lower. Furthermore, water temperature would become too high in summer since the FRG has a rule that temperature of drinking water should not rise above 15°C.

ADMINISTRATIVE CONTROL MEASURES

The hitherto described control methods deal with the management of water quality of tributaries or of the reservoir itself. Generally speaking, this depends on the structure of the catchment area. As a consequence, within the catchment area of German reservoirs protective zones can be legislated to restrict various activities (Bernhardt, 1975). These restrictions refer to housebuilding, fertilizing, transportation and storage of fuel, and so forth. The restrictions are severest close to the water edge. The public is not even allowed to approach the shoreline of drinking water reservoirs. The use of these drinking water reservoirs for ecreation is completely forbidden. The only exception n some reservoirs is sport-fishing from the shore by a imited number of people who have to renew their ishing permit every year. In cooperation with forest xperts the German water managers have worked out ules for the afforrestation of the slopes around eservoirs (Bernhardt, in press). Coniferous trees are avored to protect the reservoirs from leaf litter.

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THE EFFICACY OF WEED HARVESTING FOR LAKE RESTORATION

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ABSTRACT

Harvesting macrophytes is a useful means of restoring eutrophied lakes to a less nutrient-rich status only where nutrient loading is reduced to a low level. Submerged macrophyte biomass yield of 50 to 400 g dry wt/m^2/yr in northern lakes and 150 to 650 g dry wt/m^2/yr in southern lakes and phosphorus content of the plants of from 0.2 to 0.4 percent of dry weight limit net phosphorus removal. Harvest of plants such as water hyacinths which gain their carbon dioxide from the air may offer the opportunity for net phosphorus removal from some lakes. Macrophyte harvest is a sound ecological means of managing macrophyte abundance for recreational and aesthetic purposes and may limit internal phosphorus loading in some shallow lakes.

INTRODUCTION

An increase in the rate of phosphorus addition to most lakes is accompanied by an increase in biomass of one or more of the aquatic plant forms. The division of this increased photosynthetic activity between phytoplankton, filamentous algal periphyton, and macrophytes is determined by a complex interaction between a great many factors peculiar to the individual lake, including lake morphology and hydrology; clarity, alkalinity, and nitrogen content of the lake water; type of benthic sediment and the remainder of the biotic community. But the propensity phosphorus has to sorb on lake bottom sediments allows an early increase in phosphorus available to rooted macrophytes in lakes challenged by increased nutrient loading. Continued phosphorus loading of the benthic sediments permits rooted macrophytes to expand throughout the littoral zone. Eventual equilibrium saturation of the benthic sediments increases phosphorus concentrations in the lake water sufficient to produce planktonic algal blooms throughout the lake.

The early burgeoning growth of macrophytes responding to increased sediment phosphorus concentration in the shallow waters of lakes is a harbinger of problems often accompanied by public desire to do something about them. Depending on the prior history of the lake, this public demand for macrophyte control may occur when as little as 1 percent of the total lake area is infested with macrophytes.

Direct harvest and removal of the offending plant mass is one solution to public demand for macrophyte control which, in addition, removes phosphorus from the lake. In fact, harvest and removal of macrophytes has been suggested as a means of restoring lakes to some former less nutrient-rich status. The purpose of this discussion is to consider the potential for lake restoration and management offered by harvest of aquatic plants.

POTENTIAL LAKE RESTORATION BY HARVEST OF AQUATIC PLANTS

Restoration of a lake to a less phosphorus-rich state by harvesting aquatic macrophytes obviously requires that the amount of phosphorus removed from the lake exceed the annual net input of phosphorus to the lake. The amount of phosphorus removed by plant harvest depends on the rate of production of harvestable aquatic plant mass, the phosphorus content of that plant mass, and the efficiency of harvesting. All parameters vary considerably from lake to lake and from plant to plant.

Production of Aquatic Plants

Early estimates of the nutrient removal potential offered by harvesting were based on observations of macrophyte abundance in wastewater ponds. Development of standing crops of *Ceratophyllum demersum* of 700 g dry wt/m^2 in 60 to 70 days in a wastewater pond led McNabb and Tierney (1972) to conclude that as many as three crops of 700 g dry wt/m^2 could be harvested in a 180-day growing season. However, even with the abundance of nitrogen, phosphorus, potassium, and other required nutrients characteristic of wastewater ponds, successive harvests of macrophytes would not be possible because of limits imposed by carbon availability (King, 1972).

Impressive amounts of carbon dioxide available from the carbonate-bicarbonate alkalinity and bacterial respiration of organic matter accrued over the winter are sufficient to produce one crop of macrophytes in wastewater ponds. Harvest of this first macrophyte crop would not leave sufficient carbon for any significant subsequent regrowth of the plants. McNabb (1976) later revised his estimate of potential harvest of macrophytes from wastewater ponds to about 400 g dry wt/m^2/yr.

Most aquatic plants appear to fix carbon by the C_3 pathway and become carbon limited at aqueous carbon dioxide concentrations about 1 μmole CO/λ . Craig (1978) noted that *Ceratophyllum demersum* had a required aqueous carbon dioxide threshold for net growth of 1.3 $\mu CO_2/\lambda$ and Liehr (1978) found that the specific net carbon fixation rate decreased and the required threshold aqueous carbon dioxide concentration increased for *C. demersum* with decreasing light. Thus, submerged macrophyte growth is limited by carbon availability even in sewage lagoons where significant carbon dioxide is produced by bacterial respiration of waste organics (Craig, 1978).

Development of planktonic algal blooms is often observed following harvest of macrophytes. These algae are able to continue carbon fixation to lower aqueous carbon dioxide concentrations than those which limit submerged macrophytes (King, in press) and the simultaneous reduction of the aqueous carbon dioxide level and depth of light penetration by the planktonic algae markedly limits the potential for regrowth of the submerged macrophytes. Emergent plants gain most of their carbon dioxide from the air and are not affected much by aqueous carbon dioxide concentrations unless they are harvested to below the water surface.

McNabb's (1976) estimate of 400 g dry wt/m^2/yr for wastewater ponds appears to be about the maximum harvest of macrophytes that can be expected from lakes in the northern United States. After an extensive literature review, Burton, et al. (1978) concluded that potential harvest of submerged macrophytes would range from 50 to 400 g dry wt/m^2/yr in lakes in the northern United States while submerged macrophyte harvest from southern United States lakes could be expected to range from 150 to 650 dry wt/m^2/yr. Emergent and floating plants which gain their carbon from the air are capable of producing much greater biomass. However, these forms generally are more abundant in very shallow water and typically do not comprise a significant harvestable component in most lakes. Therefore, in most lakes, particularly in the north, macrophyte harvest will be limited largely to submerged forms.

Nutrient Content of Plant Biomass

The content of nitrogen and phosphorus in aquatic plant tissues varies with the nutrient content of the water (Gerloff and Krumbholz, 1966; Adams, et al., 1971; McNabb and Tierney, 1972). At nutrient concentrations less than the critical value required by the plants, nutrient increases increase plant production. Nutrient additions to lakes with nutrient concentrations above the critical value for aquatic plants do not yield increased production but are accompanied by increased nutrient content of the plant tissue through "luxury" uptake and storage (Gerloff and Krumbholz, 1966; Gerloff, 1975; Wetzel, 1975).

Nutrient content of macrophytes in wastewater ponds can be as high as 1.6 percent phosphorus and 5.3 percent nitrogen (McNabb and Tierney, 1972), but in natural waters range from 0.05 to 0.75 percent phosphorus and from 1.5 to 4.3 percent nitrogen on a dry weight basis. From the literature review by Burton, et al. (1979), it appears that mean values of from 0.2 to 0.4 percent phosphorus and 2.7 to 3.0 percent nitrogen on a dry weight basis would be expected for macrophytes from most lakes.

Potential Nutrient Removal by Plant Harvest

The amount of plant nutrient which can be removed from a lake by harvest of macrophytes will depend upon the density of the macrophyte growth, the nutrient content of the macrophytes, and the percent of the total lake covered by the plants. All three of these factors vary significantly from lake to lake and each must be assessed before the potential for effective nutrient removal by macrophyte harvest can be estimated for an individual lake. Knowledge of these three parameters and the annual phosphorus loading to a lake allow calculation of the degree of net nutrient removal offered by harvest of macrophytes according to the following equation.

$$\%\ \text{Removal of Annual P Loading} = \frac{(A_p)(B)(P_B)(100)}{(P_N)(A_T)}$$

Where:

A_p = Area of lake covered by macrophytes (m^2).
B = Average biomass of plants in areas covered by plants (g dry wt/m^2/yr).
P_B = Phosphorus content of plants (g P/g dry wt.).
P_N = Net annual phosphorus loading (g P/m^2 of lake surface/year).
A_T = Total area of lake (m^2).

This equation and the assumption of a phosphorus tissue concentration of 0.3 percent dry weight for macrophytes were used to construct Figure 1 which illustrates the amount of plant biomass which must be harvested to remove an amount of phosphorus equal to net phosphorus loading as a function of the percent of the total lake area occupied by macrophytes.

With submerged macrophytes yielding an annual biomass of from 50 to 400 g dry wt/m^2/yr, it is apparent from Figure 1 that macrophyte harvest would allow a phosphorus removal equal to net loadings of only about 0.1 to 1.0 g P/m^2/yr even if the entire lake bottom was occupied by macrophytes. Since phosphorus loadings of 0.1 to 1.0 g P/m^2/yr are the lower end of those usually considered excessive and since few lakes are entirely occupied by macrophytes, macrophyte harvest by itself does not offer much hope of restoring most lakes to a less nutrient-rich status. Obviously, macrophyte harvest would be of some benefit in nutrient removal in those lakes where phosphorus input could be simultaneously reduced.

Inspection of Figure 1 indicates that a plant harvest of from 2,000 to 3,000 g dry wt/m^2/yr would be required from a significant percentage of the total lake to offer much net nutrient removal from most lakes subject to excessive phosphorus enrichment. This level of plant harvest would be allowed only from lakes where plants such as water hyacinths gained their required carbon dioxide from the air (Burton, et al. 1979).

Even in those lakes where circumstances appear to favor net nutrient removal by harvesting, success will be predicted on regrowth of the macrophytes in the following years. The effect of harvest on regrowth of many submerged species is not clear, but *Myriophyllum* regrowth can be diminished by harvest (Nichols, 1974; Neel, et al. 1973; Grinwald, 1968). In addition, abundance of submerged macrophytes may decline because of other factors. Carpenter (1979) hypothesized that invasion of *Myriophyllum spicatum* may follow a wave pattern through lakes showing explosive growth followed by a decline because of unknown variables. Such lack of predictability makes difficult any reasonably accurate estimate of continued nutrient removal by macrophyte harvest.

Thus, it appears that in most lakes macrophyte harvest by itself offers no real potential of removing sufficient nutrients to yield some less nutrient-rich status. Macrophyte harvest would be important in lake restoration only as an adjunct to other restoration measures.

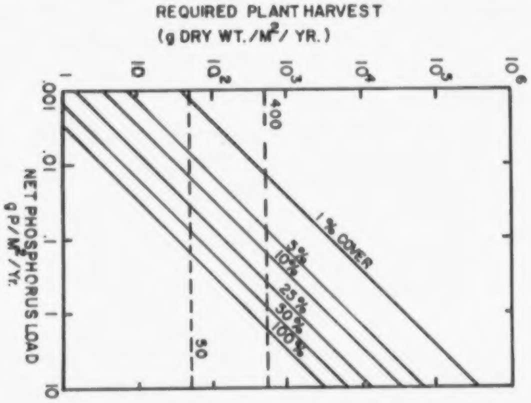

Figure 1. — Plant harvest required to equal net phosphorus loading as a function of lake area covered by harvestable macrophytes assuming a plant phosphorus content of 0.3 percent P (dry weight).

MACROPHYTE HARVEST AS A MANAGEMENT TOOL

Despite a general inefficiency in nutrient removal, macrophyte harvest offers many very real advantages in managing macrophyte abundance in lakes. It is particularly useful in controlling plant masses to allow recreational use of the lake while minimizing some of the undesirable aspects associated with complete removal of the macrophytes (Burton, et al. 1979). Removal of intact macrophytes reduces oxygen demand and the potential for winter kill of fish associated with bacterial respiration of the macrophyte biomass.

Since macrophytes can cause significant nutrient transfer from the lake bottom to the open water (Lie, 1979; Welch, et al. 1979), their harvest can help reduce internal phosphorus loading to lakes, particularly to shallow lakes. While this may reduce the recyling rate of the nutrient in the lake, the harvest itself can increase phosphorus levels in the water. The nutrients contained in macrophytes cut but not removed can be recycled rapidly to the water while the cut stems can pump at least as much as 0.43 mg P/m² into the lake water (Carpenter and Gasith, 1978).

CONCLUSIONS

The general worth of macrophyte harvest to lake management and restoration depends to a large extent on a great many variables peculiar to a given lake. Harvest offers a direct, ecologically sound control of aquatic weed abundance without adding foreign materials. In addition, it removes oxygen-demanding materials and some nutrients. As a management tool, macrophyte harvest can control macrophyte overabundance for recreational and aesthetic purposes but, by itself, is not effective for restoring most lakes to a less nutrient-rich state.

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LAKE RESTORATION — A HISTORICAL PERSPECTIVE

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ABSTRACT

The Federal Government became involved in lake restoration in the mid-1970's, but the fruits from the effort will be sampled in the 1980's, and in the years beyond. Efforts to enhance lake water quality began in some States decades before the 1970's. As we proceed with this environmental program it is well to recall the stepping stones of our progress. These began with scientific efforts by Birge and Juday to understand lakes in the early 1900's; it followed with State laws directed toward aquatic plant control of the late 1930's and early 1940's, the lengthy battle to divert sewage from the Madison, Wis., lakes in the late 1950's, the Lake Washington story, and the dedicated efforts by then Senator Mondale to enact Federal legislation to restore lakes. The present chapter in the lake saga deals with the initial Federal grant program. The program has stimulated public involvement, State lake restoration legislation, and scientific investigation. Substantial Federal, State, and other funds have been committed to lake restorative activities. The program's scientific success cannot be judged with any degree of accuracy for another decade. However, the probability for success of the lake restoration program is excellent.

We hear much in advertising these days about "a product of the 1980's, and the years beyond." Lake restoration with the involvement of the Federal government began in the mid-1970's, but the fruits from the effort will be sampled in the 1980's, and hopefully in the years beyond. Efforts to enhance lake water quality began in some States decades prior to the 1970's. As we proceed with a program to enhance this segment of the environment, it is often well to recall the stepping stones of our progress. These can be divided into investigative, legislative, and administrative actions.

INVESTIGATIVE ACTIONS

The investigation of lakes historically is anchored firmly to the comprehensive works of Birge, Juday and Smith (Birge and Juday, 1911, 1922; Juday, 1914; Smith, 1920, 1924) in Wisconsin and Forbes (1925) in Illinois. These investigators developed methods, provoked scientific interest, stimulated students, and with their prolific writings enriched the scientific literature. Even after seven decades, students of lakes would do well to examine the early writings of that era.

The stepping stones of our investigative progress have been compiled, from time to time, in several notable books or proceedings. These publications contain scientific facts and other information that remain germane today. Perhaps the first useful reference that addressed lake conditions is "The Microscopy of Drinking Water" (Whipple, Fair, and Whipple, 1927). This was first copyrighted by George Chandler Whipple in 1899. "Problems of Lake Biology" was published in 1939 (Moulton). In the foreword Moulton stated that perhaps no other biological subject involves a greater variety of interrelated factors than lake biology.

A symposium on hydrobiology was held at the University of Wisconsin in 1941. In the proceedings, James G. Needham wrote with pleasure of his personal knowledge of Stephen A. Forbes, Charles A. Kofoid, and R. E. Richardson in Illinois and of Edward A. Birge and Chancey Juday of Wisconsin. Hydrobiology, he said, is an offshoot from the old maternal rootstock of natural history, with ecology as an intimate associate.

In the 1960's, many will recall the Cincinnati, Ohio, seminar on Algae and Metropolitan Wastes (U. S. Dep. Health, Edu. Welfare, 1961) and the Madison, Wis., Symposium on Eutrophication (Natl. Acad. Sci. 1969). The latter developed as a result of a "...recognition of growing concern over problems associated with eutrophication of lakes," and was held on a very hot, humid day in June 1967, coinciding with a failure in the air conditioning system at the University of Wisconsin. Six hundred people representing 11 foreign countries and the United States attended.

Noteworthy publications in the 1970's include the "Environmental Phosphorus Handbook" (Griffith, et al. 1973), a comprehensive review of lake rehabilitation techniques and experiences (Dep. Nat. Resour. 1974), and the Minneapolis, Minn. conference proceedings on lake restoration (U. S. EPA, 1979).

The convoluted story of improving the Madison, Wis. lakes entails many years of investigation and controversy prior to eventual diversion, in December 1958, of treated sewage effluent around the two lower lakes. There are written reports of algal nuisances occurring in Madison's lakes as early as 1881. In 1884, this city of 12,000 used privies, cesspools, and direct drains to the lakes to dispose of sewage. In 1894, the City Council was told that the lakes are not to be used as receptacles for sewage in the crude state. In 1897, a sewage treatment plant was constructed but it fell far short of the claim that it would produce an effluent as pure as the water of Lake Mendota, and it was abandoned in January 1901. A new treatment plant was constructed in 1902, but periods followed when the system became overloaded and new plants had to be built.

Madison's Nine Springs treatment plant was put into operation in 1928 and flowed through the Nine Springs Creek into the Yahara River, and thence into Lakes Waubesa and Kegonsa. A Clean Lakes Association was formed in 1931 to prevent pollution of the lakes. Extensive treatment with copper sulfate to control the algae, a controversial program, was instituted and continued for many years. Sawyer and Lackey were commissioned to study the problem in the early 1940's and the results of their efforts are classic in lake investigations (Sawyer and Lackey, 1943, 1944). Sawyer published his conclusions regarding nutrients and algal growths in 1947.

Following abortive legislative efforts, protestations, public hearings, State orders, and court appeals, the State of Wisconsin Supreme Court upheld the State Board of Health and the State Committee on Water Pollution, and the Sewerage District in 1951 was forced to prepare plans for effluent diversion. The Madison lakes saga was recorded by Sarles in 1961, based largely on Flannery's historical research (1949) during his graduate studies at the University of Wisconsin.

The Lake Washington, Wash., story also extends over many years (Edmondson, 1977). Starting in 1941, Lake Washington went through a period of eutrophication because of secondary sewage effluent. The initial effects of increased abundance of algae and decreased transparency coupled with predictions of the consequences of further enrichment produced considerable concern among area residents. A public vote in 1958 established the Municipality of Metropolitan Seattle with the responsibility for improving sewerage in the region, including diverting effluent from Lake Washington. The amount of effluent entering the lake progressively decreased from 1963 to 1968. With the first diversion of about one-third of the effluent, the lake stopped deteriorating, and with further diversion it began to recover, as measured by increasing transparency and decreasing amounts of phytoplankton. By 1972 the lake began to come into equilibrium with its new circumstances.

These, and many other studies, were placed in perspective by Vollenweider (1968) in his excellent effort to determine loading rates that may be associated with biotic problems.

The phosphorus-in-detergents issue arose in 1971. On September 15 of that year, the Council on Environmental Quality, the Department of Health, Education, and Welfare, and the Environmental Protection Agency issued a joint news release on the subject. The release made four principal statements: (1) Nitrilotriacetic acid should not be used in detergents; (2) the health hazards of using highly caustic substitutes for phosphates in laundry detergents is a serious concern; (3) States and localities should reconsider laws and policies which unduly restrict the use of phosphates in detergents; and (4) EPA will begin an intensive study to identify those water bodies with a potential or actual eutrophication problem caused by phosphates. EPA also pledged assistance to States and local governments in reducing phosphates through the treatment of municipal wastes.

Subsequently, on October 27, 1971, Russell E. Train, Chairman, Council on Environmental Quality, in testimony before the Subcommittee on Conservation and Natural Resources of the House Government Operations Committee stated that the principal strategy in controlling eutrophication would be through adequate waste treatment. Two days later, William D. Ruckelshaus, Administrator, Environmental Protection Agency, in testimony before the same Committee, reaffirmed the initiation of a comprehensive National Eutrophication Survey to identify those lakes where municipal waste treatment plants should install phosphate control equipment, or where industrial nutrient sources should be controlled through the Refuse Act permit program.

On May 7, 1972, EPA announced plans to use specialized Army aircraft to collect samples "in a project beginning this month" to study eutrophication in lakes and impoundments. EPA said the survey would provide appropriate knowledge about whether a lake could be improved by reducing municipal phosphates.

LEGISLATIVE ACTIONS

Although State legislation to financially support restoration of particular lakes is of recent origin, some States have had legislation for many years to control aquatic nuisances. Wisconsin was one of the first. In 1941, the Wisconsin legislature passed an act (— 144.025 (2) (i)) calling upon the Committee on Water Pollution to supervise chemical treatment of waters to suppress algae, aquatic weeds, swimmer's itch, and other nuisance-producing plants and organisms. The Committee was authorized to purchase equipment and to charge for its use and for any services performed in such work. This cost is covered by representative taxation in a town's sanitary district.

On October 18, 1972, P. L. 92-500 was enacted. Section 314 on clean lakes (33 USC 1324) provides the legislative framework for the lake restoration program. On February 5, 1980, cooperative agreements for protecting and restoring publicly owned freshwater lakes were published as a final rule (40 CFR 35.1600).

Clean lakes section 314 had a lengthy gestation period. In 1966, Senator Mondale from Minnesota with the support of Senators Burdick from North Dakota, Douglas from Illinois, and Nelson from Wisconsin introduced Senate Bill 3769, the Clean Lakes Act of 1966 (Congress. Rec. 1966). This bill would have authorized the Secretary of the Interior to award grants and contracts to State or local agencies for comprehensive pilot programs to improve and revitalize the Nation's lakes by controlling pollution. In introducing this bill, Senator Mondale stated that minimal attention had been given to lake pollution and that there was no Federal assistance program to help States clean polluted lakes. This initial thrust was designed principally to finance pilot projects.

Virtually the same bill was introduced to the Congress again on March 21, 1967 (Congress. Rec. 1967). This bill, S. 1341, passed the Senate but did not get through the House Committee on Public Works (Congress. Rec. 1968).

On October 8, 1969, the Senate passed Senate Bill 7, the basis for the Water Quality Improvement Act of 1970. It contained provisions for basic research into the cause, cure, and prevention of lake pollution.

Senator Mondale introduced the Clean Lakes Act of 1970 on April 8, 1970 (Congress. Rec. 1970a). He called this Act an extension of the clean lakes research provisions that he had introduced in 1966. This new clean lakes bill would establish a coordinated program of increased waste treatment and lake cleaning using the latest technology to rehabilitate those lakes in particularly poor condition. Twenty-six senators co-sponsored this legislation (Congress. Rec. 1970b). This bill was reintroduced into the Senate on February 26, 1971, as the Clean Lakes Act of 1971 (Congress. Rec. 1971) The full Senate Public Works Committee voted October 18 to include the bill as part of the 1971 Federal Water Pollution Control Act Amendments. On October 18, 1972, the lake restoration section 314 became a reality in the Federal Water Pollution Control Act Amendment of 1972.

ADMINISTRATIVE ACTIONS

Particularly in the Great Lakes States, Iowa, Michigan, Minnesota, and Wisconsin, there was early concern about the growth and control of algae and vascular aquatic plant nuisances. In Wisconsin during 1949, 78 chemical treatment projects were supervised. This total included 33 projects to eradicate aquatic vegetation, eight to control swimmer's itch, five to reduce bacteria in bathing beaches, and one to spray DDT over water. This was 22 percent more than projects completed in 1948 (Mackenthun, 1950).

State-owned equipment purchased in 1941 was operated on a rental basis to chemically control aquatic nuisances. The program's early growth was governed by the ability of the operating crew to treat as many of the proposed projects as possible. It was soon found that this procedure could not keep abreast of the demands. Therefore, in 1949, the use of State-owned equipment was discontinued. Sponsoring organizations were given the opportunity to select one of two options in conducting the work: they might enter into private contract with a commercial operator or they might apply the chemical by using their own equipment (Mackenthun, 1958).

Federal interest in lake restoration began late in 1971. A relatively small discretionary grant fund was established to be used for adding a nutrient removal capability to wastewater treatment plants. A specific requirement of using such funds was that the addition of such capability was necessary to prevent eutrophication of a freshwater lake.

Controversy surrounded the National Eutrophication Survey, which was initiated prior to the enactment of P.L. 92-500. An initial requirement in selecting the lakes for study in this program was that they be receiving waters for effluents from a municipal sewage treatment plant within 25 miles of the lake. Thus, a primary purpose of the survey was to develop a need for a point source phosphorus control program. There was concern that such a program was not consistent with the needs of the Clean Lakes Program under section 314. Later, for those lakes surveyed west of the Mississippi River, the lake selection criteria were broadened to include nonpoint source and other lake problems.

Initially, also, the identification and classification of all publicly owned lakes was considered to be a problem for States to solve without Federal financial assistance. An amendment in P. L. 95-217 clarified this issue and directed the Administrator of EPA to provide financial assistance to States to prepare the identification and classification surveys required in section 314. Such assistance, subsequently, was provided in the form of matching funds on July 10, 1978 (43 F.R. 29617).

The Environmental Protection Agency did not request budgetary support for a Clean Lakes Program until fiscal year 1979. EPA believed that other programs, which required full use of available personnel had a higher priority in the goal for environmental improvement. Prior to this budgetary request, the Congress appropriated as an add-on, $4 million in FY 1975, $15 million in FY 1976, $4 million in the transition quarter, $15 million in FY 1977, and $2.3 million in 1978. Largely through the persistent encouragement of then Senator Mondale, it was possible to award the first lake restoration grant in January 1976.

THE FUTURE

There was concern within the Federal bureaucracy during the program's formative years that the Clean Lakes Program might parrot the construction grants program in eventual magnitude. I believe that fear now is abated.

The Clean Lakes Program was founded on a sound technical base. Federal funding has so far been sustained as an identifiable, consistent entity. The need for such a program is becoming increasingly apparent to a larger sector of the population. Public participation is increasing and is coming from a broader base of constituents. The future, I believe, is bright for a sustaining Clean Lakes Program.

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BENEFITS AND PROBLEMS OF EUTROPHICATION CONTROL

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ABSTRACT

The objectives of eutrophication control are commonly thought of in terms of limnological considerations, such as turbidity, algal biomass, nutrient concentrations and so forth. However, management of water resources varies greatly among different geographic areas as a direct result of different uses of the water body, as well as the historical perspectives of the user of the water. In one area, a water body considered to be enriched in nutrients could in another area be considered to be underproductive with respect to fish yields, or alternatively to be relatively pristine. It has been determined, for example, that in the Laurentian Great Lakes the trophic conditions of some of these lakes will be improved, while other lakes will be maintained simply in their present oligotrophic condition, as a result of recommended phosphorus control objectives. Some estimates of the costs and the anticipated limnological benefits of such an approach will be discussed. Socioeconomic factors, such as shoreline property values, municipal water supply costs and fish production, will also be considered, even though these factors were largely ignored in determining the recommended phosphorus loading objectives for these lakes. Comparisons will be made with other water bodies where conditions differ greatly, to demonstrate the need to look beyond limnological concerns alone when attempting to address eutrophication problems.

INTRODUCTION

Both the lake manager and the scientist concerned with water quality should reflect on why they consider lake management and restoration to be worthwhile. The purpose of such efforts must be the preservation and/or restoration of a natural resource for the greatest benefits to the greatest number of people, including scientists, managers, and the public. This paper focuses on the need to integrate considerations of the public's use of water and its perception of water quality, as well as scientific/limnological considerations, in developing effective lake management programs to control eutrophication.

Despite increasing emphasis in many countries on other environmental problems, such as acid rain and toxic and hazardous substances, controlling eutrophication is still an important issue of world-wide concern. The scientist is usually concerned primarily with the chemical, physical, and biological aspects of these problems. The lay person, while vaguely appreciating the technical aspects of water pollution, probably assesses water quality most often on its aesthetic value.

Traditionally, impacts of eutrophication have been assessed almost solely on the basis of limnological considerations. Commonly-used limnological indicators include in-lake phosphorus concentrations, Secchi depth (a measure of water clarity), algal biomass (often expressed as chlorophyll), and hypolimnetic oxygen depletion. These and other variables have been discussed by Sawyer (1947), Sakamoto (1966), Vollenweider (1968), Lee (1971), Burns and Ross (1972), Dillon and Rigler (1974), and Schindler and Fee (1974), to mention just a few authors.

As a result of this initial scientific emphasis, eutrophication control has been based primarily upon the views of scientists and engineers, with little public input or scrutiny. Consequently, water quality may be better or worse, depending on the circumstances, than the public might desire on the basis of aesthetic or economic concerns.

Public perception of water quality and eutrophication control is not, however, characterized by the rigorous analysis and review typical of the scientific view. Rather, public perception of desirable water quality can vary considerably, and is dependent primarily upon the intended use of the water and its historical perspective. For example, most North Americans look upon lakes as a focus of recreational activities, such as fishing, swimming, and boating, and as objects of intrinsic beauty. Other uses exist, of course, including commericial fishing, shipping, water supply, and waste assimilation. Obviously, not all these uses are complementary, nor are they necessarily mutually exclusive. By contrast, in less-developed countries, the production of fish as a food supply may supersede all other uses. Thus, a clear, oligotrophic lake, which produces a limited quantity of fish, would be considered an unproductive expanse of water.

Because of its technical and often complex nature, the limnology of a water body may be unknown — or unintelligible — to the public. When public perceptions of desirable water quality conflict with scientific opinion, it is necessary to recognize these differences and educate the general public as to their significance.

Therefore, although global relationships exist which describe relative trophic conditions and nutrient load-response relationships within water bodies (e.g., OECD International Cooperative Programme on the Monitoring of Inland Waters), effective lake management must not be restricted to limnological considerations, but should also include public desires. In the western world, few people would quarrel with the need for phosphorus control for eutrophic water bodies. There are frequently differences in opinion, however, regarding the in-lake conditions necessitating phosphorus control and/or to what degree such control is required. This illustrates the need for integrating both scientific and public opinions as to desirable water quality in setting goals for control of eutrophication.

PUBLIC BENEFITS AND PROBLEMS OF EUTROPHICATION CONTROL

Studies relating public benefits and problems to the technical control of eutrophication are scarce; they are difficult to generalize. By contrast, there is considerable literature concerning benefits and problems associated with eutrophication in a limnological sense. Several of the public benefits and problems associated with eutrophication control, as reported in the literature, are discussed here.

Beneficial Effects

Increased fertilization of a lake usually produces increased algal biomass, which in turn may increase the total fish production (Oglesby, 1977; Lee and Jones, 1980). Nevertheless, although total fish production is increased with increased fertilization, serious eutrophication of water bodies produces changes in fish species from the more desirable game fish (e.g., lake trout) to less desirable coarse fish species (e.g., perch). The point at which increased production of desirable game species is surpassed by the increased production of less desirable fish species, however, is not clear.

Alternatively, in situations where food supply is of primary concern, gross fish production is the most important factor, both as a social and economical goal.

As one additional benefit occasionally cited, Lefevre (1964) has reported that some algal species in eutrophic lakes have been found to produce active substances of therapeutic value for treating patients with ulcers and patients with atomic wounds.

Adverse Effects

The adverse effects of eutrophication, by contrast, have received much greater attention in the literature than the beneficial effects. That is due in part, of course, to the fact that from the perspective of man's use of water, there are usually more adverse than beneficial effects.

Although not a general concern, human health can be affected by eutrophication (Landner, 1976), though the effects tend to be chronic and are often not apparent because of inexperience on the part of physicians in recognizing the toxic effects of causative algae. Toxic effects of algae on humans may be classified as gastrointestinal, respiratory, and dermatological. As cited in Landner (1976), Dillenberg and Dehnel (1960) and Senior (1960) have reported cases of severe diarrhea, vomiting, and other discomfort occurring after swimming and/or swallowing water heavily infested with *Microcystis* and *Anabaena* in hypereutrophic lakes in western Canada. Heise (1951), also cited in Landner (1976), noted respiratory impairments in bathers in waters infested with *Oscillatoriae*; Cohen and Leif (1953) reported dermatological problems in bathers following their swimming in a lake with *Anabaena* blooms. Toxic effects resulting from algal populations (mainly *Aphanizomenon*) have also been noted for fish, poultry, and horses.

Welch (1978), citing a 1967 survey of State sanitary engineers, noted 56 percent of the total municipal surface water supplies in the United States experienced water treatment problems related to eutrophication. Cleveland water treatment plants (south shore of central Lake Erie) are frequently subjected to excessive clogging of their sand filters as a result of excessive quantities of algae in the intake waters. Taste and odor problems have also been noted in municipal water supplies (Am. Water Works Assoc., 1966). Similar problems occur in highly eutrophic embayments and nearshore areas of the Great Lakes.

Welch (1978) also notes impacts of eutrophication on industrial water supplies, shorefront property values, commercial fisheries, and recreational activities, citing several studies on the Great Lakes. The relationships between the eutrophication process in a water body and its effects on the use of the water are often ambiguous. While eutrophication control measures have often been related to improving water quality, few attempts have been made to relate these measures to their effects on the use of the water. To illustrate possible relationships between eutrophication control measures and public benefits, three specific examples are considered. These three cases, involving the Canadian portion of the Great Lakes Basin, are intended as examples of public versus limnological benefits, rather than as comprehensive analyses of eutrophication control.

PUBLIC BENEFITS OF EUTROPHICATION CONTROL IN THE GREAT LAKES BASIN

The Governments of Canada and the United States have agreed that eutrophication control through reduced total phosphorus loads is necessary in the Great Lakes Basin. Phosphorus loading objectives have been proposed for each of the lakes or sub-basins (U.S. Dep. State, 1978). The proposed target loads for Lakes Erie and Ontario and Saginaw Bay are 11,000, 7,000 and 440 metric tons/yr, respectively, all being substantial reductions from present loads. As noted in the 1978 Great Lakes Water Quality Agreement, these loads can be achieved through a combination of point and nonpoint source control measures. By contrast, the target loads for the other lakes require minimal additional reductions in phosphorus loads. The goals of the proposed phosphorus load reductions for Saginaw Bay and for Lakes Erie and Ontario are outlined in Table 1. It is noted these goals are expressed entirely in limnological terms, except for Saginaw Bay, with no attempt to relate them to associated public benefits.

Table 1. — Goals of proposed phosphorus target loads for Saginaw Bay, Lake Erie and Lake Ontario.

Water Body	Phosphorus Control Goal
Saginaw Bay (440 metric tons/yr)	Reduce filter-clogging and taste and odor problems in drinking water by maintaining an average annual total phosphorus concentration of 15 μg/l in the inner bay.
Lake Erie (11,000 metric tons/yr)	Reduce anoxia by approximately 90 percent in the central basin. Prevent any substantial release of phosphorus from the sediments.
Lake Ontario (7,000 metric tons/yr)	Minimize degradation of the ecosystem by maintaining an annual average total system concentration of approximately 10 μg/l.

Source: Task Group III (1978)

The United States and Canada have already invested substantial sums of money in attempting to reduce point source phosphorus loads and other pollution problems associated with municipal sewage treatment plants in the Great Lakes Basin. Capital expenditures for construction and expansion of municpal sewage treatment plants and sewerage works from 1971 to 1978 within the Basin are summarized in Table 2. The total commitment is large in absolute terms. It is, however, not very substantial when viewed on a per capita basis. Achieving the proposed target loads will require even further public expenditures for point source and nonpoint source controls. It is the translation of these costs into public benefits which is of interest here. Commonly-cited benefits of eutrophication control:

1. Enhanced shorefront property values;
2. Enhanced recreational values;
3. Improved commercial fisheries; and
4. Reduced costs for municipal and industrial water supplies.

Three of these benefits (shorefront property values, commercial fisheries, and municipal treatment) are discussed further in an attempt to determine whether a distinct example of associated public benefit resulting from eutrophication control can be identified.

Shorefront Property Values

Omerod (1970) considered the impact of algae-fouled beaches on property values along the Canadian shorefront of Lake Erie. He compared their real estate values (average value per foot of water frontage) for three categories of algae-fouling: (1) no algal cover; (2) light algal cover; and (3) heavy algal cover. Omerod determined that statistically there was no significant difference between shorefront property values with either light or heavy algal cover. However, the combined light and heavy algal-fouled frontage exhibited property values 15 to 20 percent below those of the shorefront areas with no algal cover.

Table 2. — Funds committed for municipal wastewater treatment plant construction in the Great Lakes Basin (millions of dollars).

Year		Canada	United States
1971		57	370
1972		66	313
1773		138	419
1974		103	509
1975		112	950
1976		174	429
1977		150	716
1978		191	618
Total		991	4324
$ per capita	(1975)	144	146

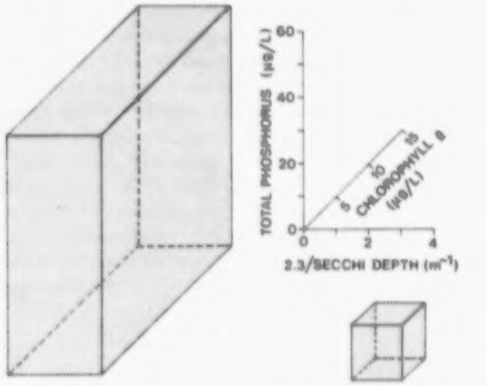
Source: International Joint Commission (1979)

A recent study by Sudar (1980) compared property values for the Bay of Quinte with contiguous eastern Lake Ontario shorelines. This comparison was based on sales information for 1971-73, converted to 1973 dollars. The median sale price per metre of shorefront for the Bay of Quinte and for eastern Lake Ontario were $495 and $449, respectively. As noted in Figure 1, however, Lake Ontario water quality, measured in terms of total phosphorus, chlorophyll *a*, and Secchi depth, is considerably better than that observed in the Bay of Quinte. These data demonstrate that factors other than water quality must account for these differences in shorefront property values. It is likely, for example, that high lake levels during the mid-1970's, and accompanying shore erosion and flooding, had a greater impact on shorefront property values than water quality degradation alone. It would appear, at least in this instance, that better water quality does not necessarily translate into higher shorefront property values.

Commercial Fisheries

Considerable literature exists concerning the impacts of various cultural stresses on the commercial fisheries of Lake Erie. One is cultural eutrophication, with its associated hypolimnetic oxygen demand and potential anoxic conditions in the central basin hypolimnion. This concern was expressed in both the 1972 and 1978 Canada-United States Great Lakes Water Quality Agreements (U.S. Dep. State, 1972, 1978). Dobson and Gilbertson (1971) suggested that the critical hypolimnetic oxygen depletion rate producing anoxia in the central basin was reached about 1960. It is also noted that major coldwater species such as lake trout, lake sturgeon, lake whitefish, and lake herring disappeared as a component of the commercial catches between the years 1940 and 1960 (Regier and Hartman, 1973; Christie, 1974).

	BAY OF QUINTE	EASTERN LAKE ONTARIO
TOTAL FRONTAGE SOLD (m)	8,128	3,009
NUMBER OF SALES	227	91
MEDIAN SALE PRICE ($) PER METRE OF FRONTAGE	495	449



PROPERTY VALUES BASED ON 1971, 1972 & 1973 SALES CONVERTED TO 1973 DOLLARS - WATER QUALITY DATA ARE SUMMER, SURFACE DATA COMBINED FOR 1972 & 1973

Figure 1

However, other stresses, including over-fishing, parasitism and/or predation by lamprey eels and other exotic species, and loss of habitat along the shoreline and in spawning streams, have also been cited by Regier and Hartman and Christie as factors affecting the commercial fishery of Lake Erie. It is not clear, therefore, that reducing the Lake Erie phosphorus load to the target level will insure the return of a viable coldwater fishery. A more recent evaluation of Lake Erie hypolimnetic oxygen data by Charlton (1980) suggests that historic increases in the apparent hypolimnetic oxygen depletion rate were not as great as formerly believed and, furthermore, that the differences which did occur in the oxygen status of the hypolimnion were more related to variations in the thickness of the hypolimnion than to changes in the Lake Erie phosphorus loads.

This example illustrates two factors which are very important in managing phosphorus loads to Lake Erie. First, the scientific information is subject to different interpretations and may, in fact, suggest a public benefit which will not necessarily occur. Second, commercial fish production in Lake Erie, the most eutrophic of the Great Lakes, is the highest in the Great Lakes system (Figure 2). Thus, if the Lake Erie phosphorus load is reduced, and results in decreased productivity, the impact upon the Lake Erie commercial fishing industry may not be positive from the point of view of the commercial fisherman. This is an instance in which public benefit and, in particular, user-specific concerns (i.e., commercial fishery) may be of more importance in the lake management decisionmaking process concerning phosphorus control than limnological concerns alone.

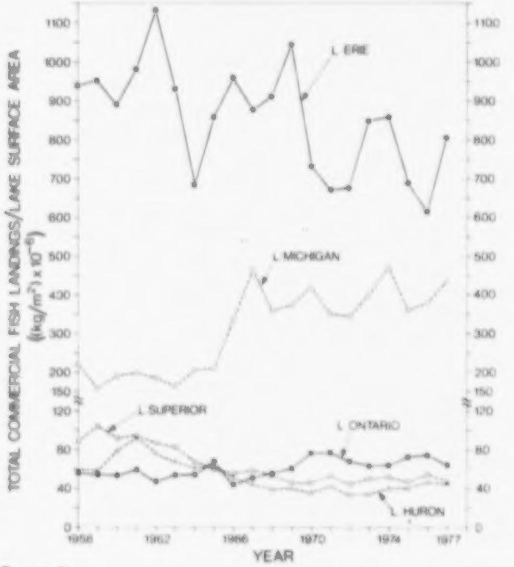


Figure 2

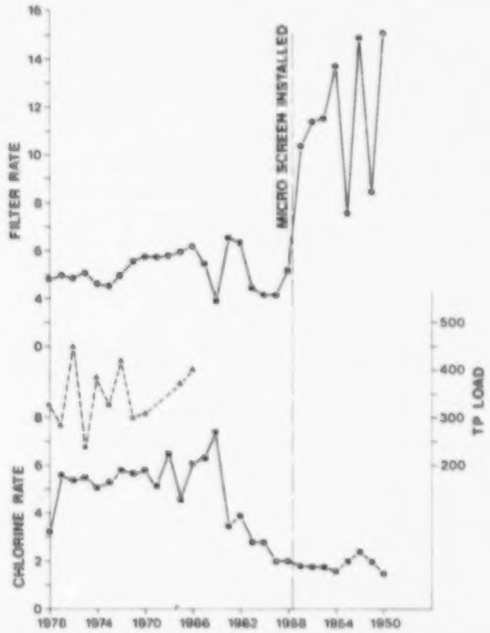


Figure 3

Municipal Water Supplies

Data for the city of Belleville water treatment plant on the eutrophic Bay of Quinte (eastern end of Lake Ontario) for the period 1950-1978 are being evaluated (Gregor, 1980) as part of an attempt to assess the impacts of eutrophication on the plant's operation. The average filter-clogging rate for the month of July for 29 years of data, based on a preliminary analysis, as well as the chlorination rate, are presented in Figure 3.

Available data for total phosphorus loads to the Bay of Quinte are also provided. Plant records indicate that filter-clogging by algae had become such a problem by 1958 that a micro-screen to strain large algae from water intakes had to be installed at the plant. The impact of the micro-screen is apparent in Figure 3. A slight downward trend is noted from about the mid-1960's to 1978. Nevertheless, it is noted that the 1978 results were obtained without the use of the micro-screen, which had been operated every summer prior to 1978. The chlorine application rate curve tends to parallel the filter-clogging rate curve, although changes in operating policy likely account for much of the upward trend in the early years. Interestingly, the 1978 chlorination rate dropped considerably, even though the micro-screen was not used that year.

At present, these data are inconclusive and warrant further evaluation. They do indicate, however, that eutrophication control efforts are likely to produce improvements in water treatment plant operations.

PUBLIC PERCEPTION OF GREAT LAKES WATER QUALITY

It is not yet possible in the Great Lakes Basin to demonstrate clearly and quantitatively the public benefits to be expected from eutrophication control. It is interesting, nevertheless, to note the public's general perceptions of Great Lakes water quality. A study, summarized by the International Joint Commission (1978), indicated that 38 percent of the people in southern Ontario, Canada, used the Great Lakes for diverse leisure activities. Perceptions of water quality trends by the user public for Lakes Ontario, Erie, and Huron are summarized in Table 3. It is interesting to note that Lake Erie was the only lake in the Great Lakes system perceived by a majority of the respondents to be improving, contrary to what one would expect based on examining the pollutant loads to the lakes. Other conclusions from this study were that:

1. The number of respondents who perceived that water quality was getting worse was decreasing with time;
2. More than 50 percent of the respondents were unaware of direct governmental measures to improve water quality; and
3. Most respondents were willing to have more of their tax money directed toward maintaining good water quality.

Table 3. — Public perceptions in Southern Ontario of Great Lakes water quality (1977).

Perceptions	Lake Ontario	Lake Erie	Lake Huron
	(percent of respondents*)		
Better	32	49	32
Worse	56	38	45
No change	6	5	10
Do not know	6	9	13

* respondents do not include non-user public
Source: International Joint Commission (1978)

CONCLUSION

The control of eutrophication is, without question, a worthwhile goal, especially in the more developed western nations, which use water for multiple purposes and therefore generally require better water quality. However, it is incumbent upon scientists and lake managers alike to consider the goals of eutrophication control in other than strictly technical terms. There is little point from a societal viewpoint in improving water quality if the views of the public as to desirable water quality are not considered. Conversely, if the public is not educated as to the scientific basis for phosphorus control efforts, such initiatives may be unnecessarily or irreversibly restricted. The effective management of water quality, to achieve the maximum beneficial uses consistent with limnologically desirable water quality, requires that these various, sometimes diverse viewpoints be integrated into overall eutrophication control efforts.

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THE POLITICS OF BENEFIT ESTIMATION

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ABSTRACT

Several studies of a newly created lake carried out in the late 1960's are used to review the current state of the art in economic benefit estimation. Techniques for direct user benefits and for indirect economic effects in the community have become well established. The basic concepts are willingness to pay and the household income partition of local business multipliers. These techniques applied to public water project investments have stressed the effects of intermediate products: For example, irrigation, power, property damage from floods. Final consumption products have been slighted, such as recreation, trauma from floods, and amenity values. Likewise, when indirect economic effects are considered, they stress employment and money income. Rarely is there a consideration of option values and/or income redistribution. Such a mix in evaluation fits the politics of traditional water resource development projects, i.e., distributive politics. But this does not fit the politics of many environmental problems which are redistributive or regulatory in character. Evaluation plays an important role in achieving political agreement. Different politics call for different evaluation techniques. Those benefits slighted by traditional analysis must be developed in evaluation if environmental restoration is to be achieved with less conflict.

INTRODUCTION

Water attracts people. A lake can be a joy to a community. Its utilitarian values are an endless list. It can perform them all while still serving as a magnet for recreation and refreshment. Just going by it every day can be a reminder of things enjoyed, of a well ordered life and environment.

At Cornell University we have Cayuga and the other Finger Lakes at our doorstep. Cayuga must be one of the most studied lakes in the world. Besides the many, many scientific investigations reposing in the libraries of the university, it has been studied by virtually every kind of resource management and planning program that Federal, State, and local government agencies can devise. A few years ago Cornell's Environmental Research Center completed an educational and research effort on lake management that concentrated on Cayuga and Owasco Lakes among others. This paper is based in part upon that experience as well as an earlier series of studies of a nearby reservoir in the Susquehanna River Basin at Whitney Point, N.Y.

Lakes are parts of larger hydrologic systems and must be managed as a part of those systems. They enjoy a distinction from the rest of the system in that they are much more easily noticed. Many people won't know which way the water in a lake flows, but they will know it's there and are more likely to be aware of its condition than surrounding streams. Thus, a lake is more likely to receive management by the public even though it may not come under a separate management entity.

Public management of such a resource includes a number of actions that are more effective the more the system to be managed is understood. But who must understand what? Many public expenditures are involved. Rules governing both public and private activities have to be devised to protect and enhance the values of the system. Monitoring, evaluation, and administrative decisions will proceed. Through it all the people involved must learn to understand and respond to the opportunities for system management. Without public management users will add their demands to a natural system until carrying capacity has been exceeded and diminishing returns set in. Investments to increase capacity likewise are limited and finally demand must be managed. Not the least of the process is accommodating water related values to nonwater interests. Water systems interact so extensively that public management means multiple agency, intergovernmental management. The result of this is that if an agency is put in charge of a natural system it never has enough control and must influence others by many means. Also, some systems are managed well enough without someone obviously in charge.

Public management decisions by definition involve politics — the balancing of a variety of interests through the structure and processes of government. We academics are quite used to the idea that decisions can be grouped by kinds of information required. But we are less aware that different decisions call forth different kinds of politics, even different sets of decisions. Public management of a water system requires a planner who in his official capacity must serve as a broker between the major interests with a direct stake in that system. The planner may be in the U.S. Corps of Engineers examining water level control options, or a consulting engineer designing a sewage treatment plant, or a regional official trying to deal with an aquatic weed problem. Whatever they try to do, they must deal with a series of veto points in their decision processes; some of these will turn out to be bargaining arenas where other interests, compatible or conflicting, will have to be accommodated or be accommodating.

The intent of this paper is to explore the differences in decisions, information, and politics and how they may be changing. In particular, the focus is on the problem of estimating benefits of public management options. Not only is the problem one of estimating *what* those benefits are, but it appears that it will be increasingly important to examine *whose* benefits are at stake.

BENEFIT ESTIMATION AND DISTRIBUTIVE POLITICS

Comparing benefits and costs began in earnest after the Flood Control Act of 1936 called for their display in any proposal for a federally funded water project. The dominant politics is that of the local public works project largely funded by nonlocal government. Here benefit to cost evaluations serve as a screen to deny or delay the allocation of the public benefit to otherwise deserving recipients. It makes manageable the competition for limited public funds, and does it on the basis of a measure of general welfare akin to the economist's concept of efficiency and gross national product. As carried out by the agencies, the concept is sufficiently flexible that it also structures support at the local level for the project. Traditionally, it has minimized conflict at both the local and national level.

Distributive politics is a term coined by Lowi to describe the classification into which most water projects seem to fall. A large number of intensely organized interests operate with their major common interest being obtaining the government action at hand. Each project is dealt with separately, and "mutual noninterference," "log-rolling," and "pork barrel" are used to describe the coalition building processes involved. Leadership is executed by brokerage and is more likely to be expressed in the legislature or in an agency rather than by the executive. Policy is arrived at more through cooptation rather than conflict and compromise. Avoiding conflict at both local and national levels is an essential ingredient to the success of their model of politics and can lead to its change. The focus is on gaining something rather than balancing of costs and returns to different groups. Costs are so diffused as not to be perceived nor well represented. Most important, the decision on how to solve a problem — the output of the policy — is made when those with the problem first approach an agency for help.

But no real situation ever fits only one model perfectly, and public policy in water resources is no exception. A second, if not dominant, model applies, labeled redistributive politics by Lowi. This is the politics of the "rules of the game." It is expressed more through an elite such as economists or environmentalists who hold important positions in the policy process. Class is more important than group. Ideology shapes policy and choice more than the distribution of benefits and costs. In redistributive politics there are rarely more than two sides to an issue, e.g., environment vs. development; and one elite for each side, rather than many separate groups. While still a gross oversimplification, two significant elite values in the redistributive political sense have been important in water resources. The first is the pressure for a rational-analytic structure and process with its two branches — orthodoxy and objectivity in economic analysis and holisitic system management. The first branch stresses proper and comprehensive economic analysis kept close to market based values and has frequently been a vehicle for asserting executive branch authority over the process. The second branch stresses river basin studies and draws some support from some State and Federal agency program managers as a way for them to communicate.

More recently and to much more effect, the environmentalist's vigorous opposition to water development projects, and indifference, if not hostility, to waste treatment works have reshaped the informal rules for distributive politics. Ingram has detailed the problem of restricting conflict over water projects when the opponents have a local as well as a national base and where there is little that the agencies can give them. What they want is no less than a different system of politics. It is well not to lose sight of the fact that in a democracy elites and ideological values prevail only when they are widely understood and accepted. Then the elites are acting with acquiescence if not with much organized support. Obviously, the body politic will accept rules that limit largess and freedom when the limits can be justified by appeal to some higher value. The makers of the principles of redistribution are indeed the holders of the command posts as Lowi argues, but the rules-of-the-game command real adherence only when everyone expects them to be enforced. When enforcement is not expected and sometimes, even when it is, requirements for such things as benefit cost analysis or environmental impact statements will be honored symbolically rather than as a substantive part of decisionmaking.

The rational-analytic model in both its economist and planner versions, with embellishments and additions from the environmentalist including a preference for demand management (conservation) and nonstructural alternatives takes second place to distributive politics.

THE BENEFITS FROM WATER RESOURCE MANAGEMENT

Benefits are estimated, in part, to be compared to each other and to the costs involved in creating them. Fair comparability can be an elusive goal, but comprehensiveness is even more difficult to achieve. Since, unlike toothpaste, there are no open markets for water services, indirect measures of direct benefits must be devised. To be comparable they must approximate the values an open market would assign. An example long used and widely accepted is the case of flood control benefits. Repair costs likely to be avoided by flood control offer a measure of what the beneficiaries should be willing to pay for flood protection. Reasonable people should be able to agree on what beneficiaries should be willing to pay if data are available to estimate likely damages.

Methodology is more easily developed and used to estimate the value of water used to produce products that are in turn sold in a market. These values are then derived from observable prices. If the shipper didn't use the waterway or the power company didn't use the hydropower, or the farmer didn't irrigate, what would they do as an alternative? The "with and without" situations are estimated and the differences in net returns are struck. Money returns are easily estimated, debated, and decided. They represent a common denomination in the analysis and help identify self interest in a decision that will affect those returns.

Evaluations that stress flood control, irrigation, power, and navigation fit the traditional expectations for water development programs and some of the public interest in them. As long as the conflicts were the interests of upstream and downstream water users, these benefits served the purposes of evaluation. But other interests have been seriously involved at least since "Earth Day" in 1970.

The addition of recreation to the recognized benefits of water development has stimulated other questions. Recreation is a product of water that is directly used. Early efforts to simply use the associated expenditures of recreationists as a proxy for the benefits created were soon disparaged. Those were costs analogous to the costs of harvesting an irrigated crop, and there was no logical reason to expect them to approximate or even correlate with the size of the net benefit to those individuals after they had paid associated costs. What should the users be willing to pay for access to the water? Examining what a consumer should be willing to pay as opposed to using the water to produce something for further sale suggests a number of questions.

Consumers, except for the very rich, can be expected to require a larger compensation for giving up an experience than they feel able to pay for using it. The willingness to sell is higher than the willingness to pay because of the constraint of income. Also something used to produce a good for sale is apt to have closer substitutes. Likewise, looking to the future, the direct consumption of resources such as wilderness recreation is apt to expand in demand relative to the more commercial values of the resource, while technology is less likely to benefit its supply.

And note that the directly enjoyed values can take on some very interesting characteristics in terms of distribution between people as individuals and as part of the community. One person's enjoyment doesn't necessarily reduce another's. Indeed, even nonusers may take more pleasure in knowing that they may become users and that others are using the resource. Also, the cost of restricting use to only those who pay for it may be very high.

The net result of these characteristics is that the stake involved in the directly enjoyed use is apt to be spread over many people and is more likely to be a small part of each of their income or satisfaction with life. Commercial uses are apt to mean a great deal to a smaller number of people. Thus, the cost of getting organized to either bid for the resource or to represent their interest politically is much greater for the diffused, direct enjoyment users than it is for the commercial users. An important exception would be where this disadvantage is widely recognized and political leaders are supported in efforts to tilt the rules of the game in favor of individuals.

Extending these concepts to water quality management, fish and wildlife values, aesthetic, cultural, and spiritual values has been suggested. Also, flood control benefits based only on property damage can be seen to be deficient when the value of trauma avoided is considered. It would seem logical that the more violent, harder to deal with floods, may be more trauma producing. If true, developing valuations for flood trauma may reinforce flood plain management alternatives at the expense of structural measures for flood control.

Always a concern in methodology development is whether two or more approaches are actually measuring the same thing, and whether the instrument imparts a bias of its own. For example, when you ask people what they will pay, will they in fact do so? Probably not, but what to do instead, and how much is the bias? Will respondents shade their answers to give what they think the investigator wants, and what will that be? Will they expect to have to pay if they say they will? Or will they raise their values to induce more of what they want to be provided, since they can't imagine that they can be made to pay? The alternative to asking questions is to use costs incurred by users such as travel and time to relate differences in cost to quantity enjoyed. Such a travel cost method is employed to relate quantity to price in a manner similar to demand studies in a conventional open market.

In a study carried out at a reservoir in central New York in 1966, Romm compared 10 techniques. Table 1 summarized the results. While the range is large — four and five to one — plausible arguments suggest a part of the variation is due to differences in what is being measured. For example, travel cost has many elements that are not variable with the trip. The facility in question was very heavily used by young people and given our youth culture this may have added to the larger value for willingness to have government spend.

Table 1. — Alternative estimates of the value of recreational use of a reservoir, Whitney Point, N. Y., 1966.

Summary of method	Benefit Per User U.S. dollars (1966)
Travel cost, without time value adjustment	0.29
Additional distance willing to travel — hypothetical bids	0.35
Willingness to pay fee — hypothetical bids	0.39
Combined distance and fee — hypothetical bid	0.63
Willingness to pay in addition to taxes — open end question	0.26
Willingness to pay asked after question on government investment	0.45
How much should government spend per user day?	$1.31
Compared to next best priced alternative, recreation preferred	$1.03
Compared to priced alternative, recreation preferred and rejected	$1.35
Cost of requirements for the same activity	0.75

Source: Romm, 1969.

Bishop and Heberlein report on an experiment that created a market where one had not been before. They bought special goose hunting permits in Wisconsin (for an average of $63) and then compared that result with hypothetical offers of willingness to sell ($101), willingness to pay ($21), and travel cost estimate with no time value ($11) up to one-half the income level

($45). The relative positions of the estimates were as expected, but the magnitude of the differences — a range of ten to one — was surprising.

Batie and Shabman point out a critical problem in the development of this type of methodology. Estimates almost always value the complete, directly enjoyed service. But the planner and the policymaking process must deal with policy measures that affect less than the whole value. In other words, they have not addressed the "with and without" problem. Focusing on the control variables is surely an important challenge for research. But that is not likely to explain why planners and agency analysts have moved slowly and selectively in estimating directly enjoyed benefits. This requires an explanation that deals with the politics or institutional setting for the use of such information. The methodology of manageable costs has been available, could have been adapted to the "with and without" problem, and gives results that are no less precise than the more traditional benefit estimates for flood control, irrigation, navigation, and hydropower and reservoir recreation.

SEEDS FOR A SHIFT IN THE POLITICS OF WATER RESOURCES

Planners estimate benefits and costs (including the loss of existing benefits) to match the demands for information of the kind of politics in which they find themselves. Distributive politics requires finding local agreement and translating that into national agreement. Increasingly, conflict cannot be contained. Formalized public participation processes have been introduced partly to deal with this and partly to respond to a more recently emphasized change in the rules of the game by the holders of the command posts.

An example is the recent national water quality planning exercise under section 208 of P.L. 92-500 where various advisory committees were required: technical, public, and local government. Contracts were signed with various nontraditional water quality clientele (agricultural and forestry agencies, general regional planners) for plan elements. Farm and forestry agencies that are developing water quality programs will probably continue as part of the water quality network locally and nationally. In some localized situations, local governments have used the opportunity to develop permanent management capacity. But many communities lacked a publicly demonstrable, immediate problem; the permit process covered industrial discharges and federally funded municipal plants were controlling municipal discharges. There was little basis to devise new institutional arrangements. But what of the future? Will the very high costs of advanced treatment and the extreme system burdens posed by toxics and nonpoint pollutants provide the impetus for a regional approach? Will the rational-analytical model at the basin level be given new life because it can do the job more cheaply? If so, much of the bargaining over pollution standards and enforcement would shift to the regional level from the State-Federal focus it has enjoyed in recent years. Achieving agreement in the face of environmentalist opposition may force something similar on the dam and channel building agencies.

In water resource development, informal public participation has never been lacking. The support requirements for dam and channel projects have been substantial from the time studies begin to the last Federal dollar spent decades later. For public sewage treatment plants first-come, first-served rules and a complex process of reviews have tried the persistence of none-too-enthusiastic local officials. As Holden observed in 1966, water quality in both the permit and treatment plant construction activities have always involved a substantial amount of bargaining between polluter and enforcement official. This is a type of politics that fits neither the distributive nor the redistributive models but still a third general theory of politics which Lowi calls regulatory politics. It can apply to much more than the public activities usually designated as regulatory. Changes in both water development and water quality management may increase the significance of this type of politics and with it change the kinds of information, including benefit estimates.

The essential features of regulatory politics are captured in the pluralist tradition of political science. Policy is the result of group conflict and the groups are large and well organized. It is not the result of log rolling by many small groups who have nothing else in common but groups whose interests collide. It is not a question so much of colliding values which in our system may go forever without being resolved. Rather one group cannot continue to enjoy its values unless it can achieve an accommodation by another group. Rules for accommodation tend to be broad and give the appearance of inflexibility. Examples include "zero discharge" and "non-degradation." Subsidies are more openly identified. Leadership and coalition members may be too unstable to fit the term of an elite in the political authority sense. Bargaining, mediation, agreements, and acceptability characterize an emphasis on process. Information on the benefits and costs enjoyed by different groups might become grist for the mill rather than symbolic accommodation of a general value or ideology.

But water resources management is a localized and sectionalized phenomenon. The focus is on the lake and the watershed and the associated communities. Also for more effective future management, many of the functions jealously performed by local governments will need to be used — land use controls are a case in point. The distributive politics involved will still dominate at the national level. This suggests that the scope for expanding regulatory politics is at the local level to achieve consent and agreement that can be transmitted to the national level.

Where distributive politics are hamstrung by local conflict, the search for a broader coalition should look attractive. The key step will be in avoiding early commitment to particular means to solve problems. But this will require moving away from the presumption that Federal money will be available only for dams, channel works, sewers, and treatment plants except where mitigation and similar bargaining yield funds for fish and wildlife and recreation facilities. Broader access to alternative means will attract new support groups and encourage accommodation to environmental interests.

A planning process that emphasizes identifying more of the benefits earlier — even before they can be refined to fit the specifics of particular options — suggests that conflicts may surface while they can still be accommodated in the planning process. If no conflict arises, distributive politics can proceed as usual. If it does and no accommodation appears possible, the unsatisfied interests — whether because of deeply felt value conflicts or otherwise — will have received a more obvious application of political due process. Planners will have a better chance to display accommodations which may still not be acceptable to conflicting ideologies but which others find acceptable in their behalf. Remember the test for willingness to pay is not that the beneficiary is indeed willing, but that reasonable people agree that the beneficiary should be willing.

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CLEAN LAKES ESTIMATION SYSTEM

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ABSTRACT

In administering the Clean Lakes Program the U.S. Environmental Protection Agency, Office of Water Regulations and Standards has taken a number of steps to assure efficient management. Those steps include establishing a regulation for administration, seeking additional funds to support the Program and establishing a Program Strategy. This paper describes a part of that strategy: the Clean Lakes Estimation System and its role in the Clean Lakes Evaluation System. This combines Federal, State, and local decisionmakers with an information system which can be used to estimate the various ecological and human consequences of a 314 project and human value of all those consequences. The Estimating System being developed will provide information for use in evaluating the results of the entire Clean Lakes Program and also in choosing projects for funding. The components of the Estimation System are: (1) Procedures to estimate the limnological and other ecological outcomes of various treatments; (2) procedures to estimate the various human impacts of a 314 project; (3) procedures to identify the value of the various impacts in terms of standard economic values; (4) procedures to identify the value of all consequences in terms of stated local, State and Federal goals.

In administering the Clean Lakes Program, the U.S. Environmental Protection Agency, Office of Water Regulations and Standards, has taken steps to help assure efficient management: establishing a regulation for administration (Code Fed. Reg., 1980), seeking additional funds to support the program, and establishing a Clean Lakes Program Strategy (U.S. EPA, 1980). This paper describes an essential component of that strategy: An improvement in the Clean Lakes Evaluation System which we term the Estimation System. First is a general description of the proposed Estimation System and how it will fit into the total Evaluation System.

EVALUATION SYSTEM

The Clean Lakes Evaluation System has a dual purpose: (1) To select, in as wise a way as possible, the best restoration or protection projections for funding under section 314; and (2) to provide continuous evluation of the entire Clean Lakes Program for use in improving the Program and in assessing its benefits. The Evaluation System is used to select projects for funding by identifying project goals and forecasting the degree to which those goals will be met. Projects which promise success through high goal attainment are the best candidates for funding.

The results of the Clean Lakes Program are evaluated by ascertaining the degree of goal attainment in completed projects. Thus, the System evaluates proposals for funding by estimating the likely results. It evaluates the Program by estimating actual results.

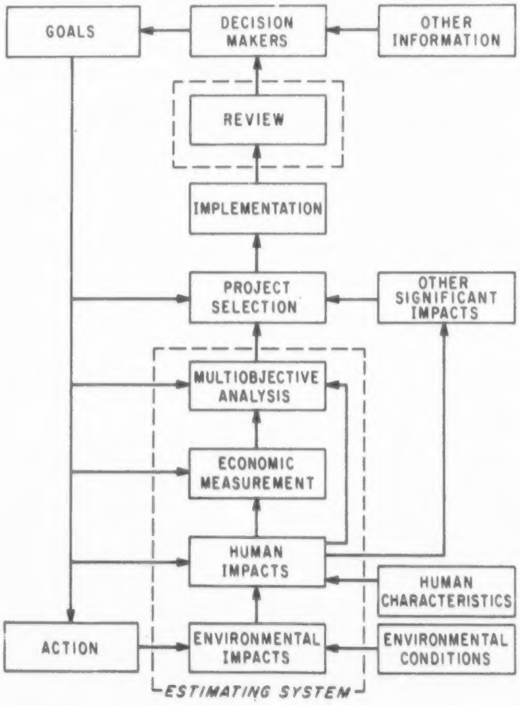

Figure 1. — Clean Lakes Evaluation System.

Figure 1 shows the elements of the Evaluation System and their interrelationships. A 314 project proposal arises because some *decisionmakers** (Figure 1: top, center) judge some current and potential *environmental conditions* (bottom, right) to be change-able and inconsistent with *goals*, i.e., undesirable. The *actions* taken to carry out the project, combined with existing environmental conditions, lead to various *environmental impacts*. The environmental impacts propagate through webs of cause and effect until they lead to some *human impacts*, including desirable changes (positive impacts) in at least some of the conditions which led to the 314 project proposal. There may also be impacts (either positive or negative) which have not been anticipated. The nature and magnitude of the human impacts are affected by the *human characteristics* involved. For example, urban dwellers often differ from their rural cousins in attitudes about natural resources.

Environmental and human impacts are often so numerous that we are forced into some degree of abstraction when identifying those to consider in the evaluation process. In principle, we choose those impacts which are of greater significance. Significance is based on the number of people impacted and the enormity of the impact on each individual. A guide to the significance of the various impacts can be found in the goals of a 314 project, for those goals led to the proposed actions.

To date, identifying goals, determining human impacts, and evaluating the positive and negative aspects to arrive at a good project design and then judging the desirability of the project has been a qualitative process, and in many cases implicit rather than explicit. Thus, in terms of Figure 1, the evaluation of a proposed project has tended to have no *economic measurement* or *multiobjective analysis*. Most impacts are of the *other significant impacts* sort and outcomes are weighed in qualitative terms. A project design is evolved and that design is analyzed to decide whether or not to *select* the *project* for funding. if so, *implementation* occurs with various results which, through *review*, provide feedback to the decisionmakers. This feedback is combined with *other information* to either reaffirm or modify goals for subsequent proposals.

ESTIMATION SYSTEM

The Clean Lakes Evaluation System is composed of two major parts. One consists of the various decisionmakers at the Federal, State, and local levels involved in designing, selecting, and implementing 314 proposals, as well as those who administer, fund, and otherwise influence the entire Clean Lakes Program. The second part is made up of the information systems used by the decisionmakers to obtain estimates of the outcomes (either expected or realized) of individual projects, or of the Program as a whole.

The Office of Water Regulations and Standards intends to make certain portions of the information systems explicit to improve both the quality of the information and the ease of its communication to various decisionmakers. This will improve the office's evaluation of proposed projects, and thereby its selection process, as well as provide a more objective basis for funding and improving the entire Clean Lakes Program. The improvements all relate to the bottom four boxes, and their associated arrows, in the center column of Figure 1 and their duplicates in review. Accordingly, the improvements will include procedures to estimate:

1. The limnological and other ecological impacts of various treatment possibilities in a 314 project.
2. The various human impacts of a project.
3. The value of the various impacts, where possible, in standard economic units, i.e., dollars.
4. The value of as many as possible of all the impacts in terms of stated local, State, and Federal goals through a multiobjective analysis.

Together these four sets of procedures are termed the Clean Lakes Estimation System, as shown in Figure 1. The Estimation System is intended to be comprehensive in two dimensions. One, which we might term the vertical dimension, includes the whole series of impacts stemming from restorative (or protective) action to ecological impact to human impact to the values of those impacts. The second, or horizontal, dimension includes all human values affected by a 314 project and the ecological and human impacts which impinge on those values. Comprehensiveness in the vertical dimension is necessary to insure that evaluations are made in terms of human values, and that only changes in these values which can be traced back through the social and ecological systems to the actions undertaken are credited to the project. Comprehensiveness in the horizontal dimension is necessary if myopia is to be avoided and the spirit of the National Environmental Policy Act (42 USC 4321) and the Principles and Standards of the Water Resources Council (Fed. Reg., 1973) and other statements regarding the management of public resources are to be followed. This is particularly necessary because many 314 projects include a wide variety of treatment activities both in a lake and throughout the watershed.

However, some impacts and values will always be unknown and some which are known will not be contained in formal estimating procedures. Therefore, the comprehensiveness of the Estimation System, like all other aspects of the Clean Lakes Program, will be continually subject to review and modification.

ESTIMATING IMPACTS

The first step in developing the Estimation System will be to develop a list of significant environmental and socioeconomic impacts which are anticipated from various types of 314 projects.

For each type of significant impact, a procedure will be developed for estimating the magnitude of that impact on the basis of certain, presumably causal, factors. The procedure will, in essence, be a series of equations in which the impacts of concern are the dependent variables and the causal factors are the independent variables. To illustrate:

suppose $S = b_0 + b_1 P - b_2 A$

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* The elements named in Figure 1 are italicized when first used in the text.

where S = number of swimmers using a lake
P = population of the community
A = concentration of algae in the lake
b_i = various coefficients developed through analysis or otherwise

To employ such an equation requires values of the independent variables (P and A). The variable A is the link back to the 314 project, for presumably A is to be reduced. The pre-implementation value A_0, leads to an estimate S_0,, and the post-implementation value, A_1, leads to an estimate S_1. Of course, the value of P may also change. The change in $S(S_1-S_0)$ is therefore one of the human impacts of the restoration.

To develop precise and cost effective estimating procedures, equations must be formulated on a regional basis with certain independent variables included which allow the prediction to be fitted to a given 314 project. Using the illustration presented earlier, let the number of swimmers be determined as follows:

$$S = b_0 + b_1P - b_2A + b_3U$$

where S, P, A and the B are defined as before, and U is a variable representing urbanization.* If city dwellers have a significantly different attitude toward swimming than rural dwellers, than U will be important; including it makes it possible to determine one equation for a whole region which can be applied with precision to a given locality.

VALUES

As described so far, the System yields estimates of the impacts of a 314 project. Some of these impacts can be validly measured in economic terms, some cannot. Some values measureable in economic terms are costs of restoration, tax impacts, property value changes, business activity, some damages caused by floods, etc. These impacts will occur in ordinary markets and so long as these markets are relatively well organized they will reflect the values involved.

Other values not explicitly identified in a market can still be measured in economic terms by virtue of their close relationship to a market. Two notable examples are benefits to recreationists and costs of modifying farm management practices. The field of recreation economics has devised a variety of tools for estimating the value of recreation based on a proxy price, such as travel cost. Modified farm management practices, if instituted on a wide scale, will ultimately be reflected in the market prices of farms. In the meantime, farm management models can be used to estimate the impacts on farm profits. Profits, like travel costs, are the market values which can be used to estimate the value of a closely related impact.

Finally, some impacts are so removed from ordinary market processes that economic valuation is not possible. Economic measures of the value of items such as community cohesion, education, and research often have little validity. However, one must be careful in making such judgments. In a specific context it may be possible to derive a valid, useful economic measurement of almost anything. Consider a human life: Can its value be expressed in economic terms? In general, I'd say no. Such approaches as discounted earnings or life insurance carried, etc., reflect only poorly and partially the contribution of a person to his child or to society. But, in certain specific situations we can measure the value society implicitly places on human life. Consider the case of plane passengers: By computing the costs of safety regulations and the number of deaths per million passenger miles we can derrive an implied value per life. This value can be useful to decisionmakers concerned about possible modifications in safety regulations, particularly when compared to other modes of transportation and other types of activity.

The boundary between values which can, and those which cannot, be correctly measured in economic terms is fuzzy since it depends upon the decision being contemplated and how the measurement will be used. This is another way of saying that a correct economic valuation procedure depends on the goals involved. The value of a human life appropriate for determining public safety regulations is quite possiblly different from one appropriate to a damage suit involving negligence. And neither is likely to adequately measure the value to one's child. For this reason, Figure 1 shows goals to be a determinant of economic measurement in addition to their role in determining the significance of human impacts. It is also true that over time, certain categories of value become measurable in economic terms as the field advances: Fifty years ago economic valuation of publicly provided recreation benefits would have been impossible.

EPA intends to identify the value of those impacts which cannot be correctly obtained in standard economic terms through a process called decision analysis by some and multiobjective analysis by others (Keeney and Raiffa, 1976). The values would be obtained through the goals and utility functions of decisionmakers. The utility functions specify the degree of goal attainment associated with various levels of some objectively measurable variables. The measurable variables are those impacts estimated by the System as described previously.

Consider the following simplified example. Suppose two goals exist for a 314 project:

1. Maximize net financial worth of recreation benefits minus project costs.
2. Maximize wildlife diversity and abundance through the use of wetlands as nutrient sinks to create new and diverse habitats.

These two goals cannot be met simultaneously since maximizing goal 2 implies a very large budget which would lead to a less than maximum value for goal 1. Assume the decisionmakers judge goal 1 to be three times as important as goal 2. Suppose the utility functions for each goal are as shown in Figure 2. If a particular project design promises a net financial worth of $1.5 million (utility = .9) and a wetlands budget of $250,000 (utility = .5) the value (V) of the project $V = 3(.9) + 1(.5) = 3.2$. This magnitude of V could be compared with that of other designs to pick the best design and with that of other projects to select the best projects for funding.

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* Urbanization is a qualitative concept. An acceptable quantification will need to be determined. Simple formulations such as population density or city size may be adequate, or more complex formulations may be needed.

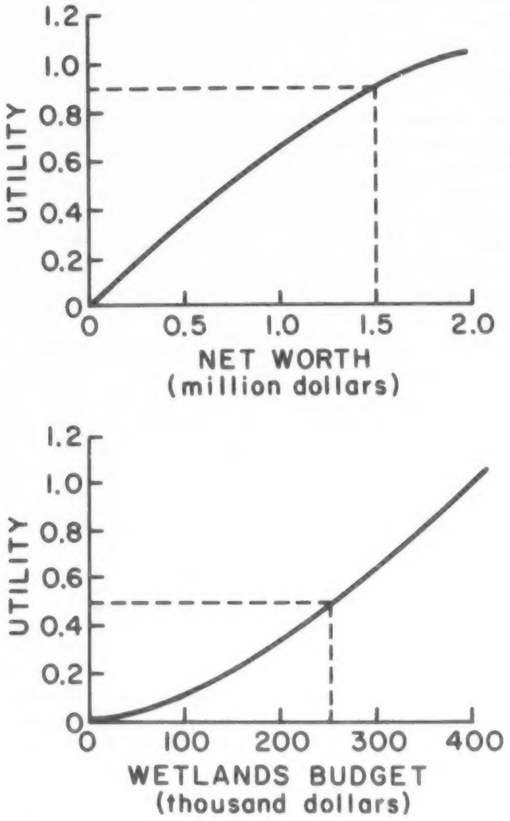

Figure 2. — Utility associated with each of two goals.

SELECTION AND REVIEW

The Estimation System can also be used any time after implementation to see the degree to which goals were actually met. The pre-implementation and post-implementations results could differ if any of three things occur: Errors in estimation, shifts in value structure, or inadequate management.

Consider the example presented earlier involving two goals: Financial worth and wildlife. Suppose the design having a value of 3.2 was implemented. Suppose upon review a value of only 2.0 resulted. Suppose further the difference in value was due to two sources: A net worth of only $1.0 million occurred instead of the $1.5 million anticipated, and the wildlife utility function fell. The lower net worth might be traced to higher than expected costs or lower than expected benefits of some specific type. The downward shift in the wildlife utility function might reflect the fact the local populace considered the wetland habitat to be less useful than they had anticipated. This, in turn, might result from a number of causes. Perhaps the area was insufficiently accessible. Perhaps the community had been initially oversold on prospective benefits and the shift in utility was a reflection of reality. Or, perhaps a public education program was needed for the community to observe and appreciate the wildlife impact.

The pre- and post-implementation values of this project could also be compared with those of a number of other projects. Do they all show a decrease between anticipated and actual value? If so, perhaps the program needs to be reduced. Do some projects show large increases and others large decreases? If so, this may indicate the presence of some important variables not presently accounted for in the Estimation System. Some such variables can be analyzed for inclusion. Others may be identified, but accounted for in only a subjective way.

One such variable is management. For instance, a successful implementation may require effective management of a lake district or the coordination of several local, State, and Federal agencies. If this effective management or the necessary institutional framework is absent, a well planned restoration may not succeed. Therefore, it will be necessary to ascertain what institutional or administrative factors are associated with successful and unsuccessful projects. Since no plan is constructed with perfect foresight, a deviation does not necessarily indict management. What counts is whether or not the final result is considered to have been worth the effort. In either case, there is something to be learned from the process which can help future implementations succeed.

It is clear the review process is exceedingly important. It can detect errors in the Estimation System, ascertain shifts in value structures, and identify important new variables, for specific attention in the evaluation process.

INFORMATION

A point worth emphasizing is that the Estimation System, as envisioned, is a set of procedures to estimate what *would* occur and what *did* occur in the ecosystem and social system, given certain conditions and actions. Therefore, the Estimation System does not make decisions, it generates data for use by decisionmakers. The System is confined to answering questions about what would (or did) happen if . . . This is obvious in the estimation of environmental and human impacts. It is equally true in economic measurement and multiobjective analysis. These last two items estimate values which occur given certain conditions and actions. Weighing the various alternatives to finally arrive at project selection or Program modification requires additional inputs. In Figure 1 these inputs are symbolized by the box "other significant impacts" which include not only human impacts and values unaccounted for in the Estimation System, but also judgments by the decisionmakers of the validity and precsion of the various components of the Estimation System.

From the illustrations given, the Estimation System may appear to be totally quantitative. This need not, and hopefully will not, be so. Estimates may just as well be qualitative as quantitative. Indeed, at any one moment in time, qualitative estimates may contain more information than is possible with currently available quantitative technology. The Estimation

System is intended to be an information generating system, not an information destroying system. Therefore, if comprehensiveness is to be obtained in the vertical and horizontal dimensions as stated earlier, the System will inevitably contain qualitative elements.

However, it is also true that since quantitative information is easier to transmit and transform than qualitative information, there will be a tendency to develop quantitative expressions wherever appropriate. This leads to three recommendations for those developing and using the Estimation System:

1. Those who develop quantitative information should be humble enough to state what their numbers do not contain. This includes error terms in the statistical sense. It also includes statements about omissions resulting from the way the problem was formulated to make it amenable to quantitative analysis.

2. Those who develop qualitative information should be concise — not quantitative, but concise — if they don't want their information treated like excess baggage.

3. Users of the Estimation System should view it with a jaundiced eye: Its output is probably in error. The need is to know where those errors lie, how significant they are, and how to compensate for them.

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IMPACTS OF LAKE PROTECTION ON A SMALL URBAN COMMUNITY

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ABSTRACT

The Waupaca City Council was the first municipal body in Wisconsin to form a lake management district to take advantage of new legislation and funding for lake cleanup. Efforts to manage Mirror and Shadow Lakes began in the 1960's when it became apparent that the two small lakes located within the city limits were experiencing water quality problems. Research revealed that excessive nutrients were entering the lakes through storm sewers. In 1975 the Waupaca Lake District requested State and Federal financial aid for a stormwater diversion project, and became one of the first awardees under the Clean Lakes Program. An evaluation grant accompanying the EPA implementation grant examined limnological, sociological, and economical impacts of such a project. The evaluation of the sociological impacts examined the effects on individuals, groups, and local government. The project generated only mild sociological benefits beyond those associated with water recreation itself. In the economic evaluation it is the critical explanatory variable, water quality, that must be accounted for in the relevant models, such as predicting the impacts on property values and the recreation response. The methods used in estimating these impacts and the interpretation of the results obtained from these models are presented in this paper.

INTRODUCTION

Under the Clean Lakes provision of Public Law 92-500, the Environmental Protection Agency embarked on a major program of cost-sharing grants to implement lake restoration projects. Since requests for financial assistance exceed available funds, an evaluation of project impact is crucial to sound decisions on future applications. Both potential grantees and EPA need to know how past efforts have fared. Project justification, optimal level of implementation, and relative priority of individual projects depend on such evaluations. Public investment decisions can best be made if potential impacts can be predicted based on systematic evaluations of the following procedures:

1. Limnological evaluation to determine whether water quality has been improved (or maintained);
2. Economic analysis to determine monetary benefits relative to investment; and
3. Sociological assessment of non-monetary impacts on individuals and groups.

In this paper we are primarily concerned with the economic and sociological impacts of the Mirror/Shadow Lake Project in Waupaca, Wis.

Waupaca

The city of Waupaca has experienced a steady population increase since 1960, exceeding the average growth rate for rural communities which reflects a nationwide shift away from urban areas (Waupaca County Outdoor Recreation Planning Committee, 1978). The current population of the city is approximately 5,000.

During the summer, the area's population more than doubles as second-home owners and vacationers immigrate to Waupaca and the surrounding chain-of-lakes region. These summer residents participate in pleasure boating, canoeing, swimming, hiking, picnicing, and scenic driving. They contribute an estimated $1,050,000 a year to the local economy (Cooper and Powers, 1976). Clearly, the maintenance of environmental quality, especially the water resources, is critical to the continued economic well-being of the city of Waupaca and surrounding areas.

Mirror/Shadow Lakes

These small lakes, of 5 and 17 hectares respectively, are located within the city of Waupaca. South Park, on the municipally-owned west shore of Shadow Lake, provides the only swimming beach in the immediate area. The park is heavily used for picnicking as well as swimming. In 1978 the attendance at South Park was estimated to be 83,809 local users, with nonresidents accounting for an additional 10 to 15 percent of that amount (Bouwes, et al. 1980). Both lakes began experiencing algae problems and dissolved oxygen depletion in the late 1960's. The quality of Shadow Lake was still acceptable, but the drainage of low quality Mirror Lake water into Shadow Lake concerned lake users.

In response to that concern, Waupaca created a lake district in 1974, the first year the Wisconsin Lake Management Law went into effect providing for such local management units (Chapter 33, Wisconsin Statutes). Studies revealed that most of the phosphorus entering the lakes could be traced to storm sewers emptying directly into them. With technical assistance from the Wisconsin Department of Natural Resources, the lake district proposed a three-phase project to deal with the problem:

1. Eliminate most of the phosphorus loading by storm sewer diversion;
2. Treat the lakes with alum to precipitate the phosphorus in the water column and seal off the phosphorus-rich sediment; and
3. Aerate Mirror Lake to promote turnover since natural turnover by wind action is inhibited by its depth (13 meters) and sheltered location.

The storm sewers were diverted in 1976, alum was added in 1977, and aeration began in 1977 at a total cost of approximately $430,000. This cost was shared by EPA (50 percent), DNR (30 percent), and the local lake district (20 percent).

LIMNOLOGICAL IMPACTS

Limnological evaluations, which have been conducted since 1977, have revealed that phosphorus levels have dropped in both lakes and oxygen levels have increased to again support fish life in Mirror Lake. Water clarity has not improved. *Oscillatoria rubescens*, a blue-green alga which lives deep in the water column, has been replaced by green algae which are characteristic of less eutrophic lakes and support a better aquatic food chain, but they also grow closer to the surface where they are more visible.

ECONOMIC IMPACTS

A thorough analysis of the economic impacts of a project should include both allocative efficiency and distributional equity considerations. The efficiency issue examines whether the reallocation of resources to the project, e.g., those used for water pollution control, increases the net value of the output produced by the resources. Ideally, one would wish to determine not only if the resources had been optimally allocated among alternative uses, but whether they are optimally allocated for a given project. The equity issue examines the welfare redistribution associated with a project; that is, the distribution of the project benefits and costs.

Efficiency

One of the common tools employed to examine project impacts is a benefit-cost analysis. Such an analysis seeks to answer the question: "Are the benefits, i.e., increases in welfare, generated by a project greater than the costs necessary to realize the project?" In the case of a project with water quality impacts, it is necessary to employ a methodology which will allow values to be imputed to water quality as a nonmarket good, since the market fails to provide prices (values) directly.

Economic theory and earlier research indicate that the benefits associated with the Mirror/Shadow Lakes improvement project will be capitalized in the surrounding property values. Consequently, a property value model was used to estimate these impacts (Dornbusch, 1974).

The basic premise of this model is that water resource projects have value to the general public which, in the absence of a market for direct sale of this output, are adequately reflected in the market prices of those properties situated near the resource.

When a public project enhances productivity or utility, benefits accrue to the affected firms and households. These benefits increase the value of certain locations, and as a result, the initial equilibrium in land markets will be perturbed. Eventually, new equilibrium land values are established. The total benefits from such a program equal the sum, over all firms and households, of the changes in productivity and utility, and, therefore, are equal to the sum of the changes in land values from the initial equilibrium to the new equilibrium.

The empirical model postulates first that a change in property values is due to *perceived* changes in water quality by area residents, and secondly, that the impact on property values decreases as distance from the lake increases. These two aspects are incorporated into the following equations:

a. $\mathrm{PWQI}_{\mathrm{Exp}} = \sum_{K=1}^{7} a_k B_{ijk}$

b. $\mathrm{PWQI}_{\mathrm{Res}} = -24.778 + 0.463\,(\mathrm{PWQI}_{\mathrm{Exp}}) + 15.5\,(\text{Public Access})$

c. $b_1 = e^{6.398}\,(\mathrm{PWQI}_{\mathrm{Res}})\ 0.492\ e^{1.180}\,(\text{WBT Lake})\ e^{0.991}\,(\text{WBT Bay})$

d. $b_0 = -b_1 \frac{1}{(DW_{max})}$

e. $\Delta P\%_d = b_0 + b_1 \frac{1}{(\overline{DW}_d)}$

Equation (a) determines the experts' perceived water quality index ($PWQI_{Exp}$) which represents a limnologist's *ex ante* estimates of how much each of seven different water quality parameters would change both with and without the project. These seven parameters are (1) industrial wastes in the water, (2) debris in or on the water, (3) clarity of the water, (4) algae in the water, (5) odor from the water, (6) wildlife support capacity of the water body, and (7) the recreational opportunities affected by the water level. B_{ijk} reflects the change from water quality condition i to water quality condition j for the kth parameter. The relative importance of each of the seven parameters is represented by the weighting factor a_k.

Equation (b) expresses the perceived water quality index rating by residents ($PWQI_{Res}$) which is a linear function of the expert's perception of water quality and the degree of public access available at the lake. Consequently, by being able to predict residents' reactions to a given water quality change, this equation provides a vital link which allows for an *ex ante* evaluation.

Equation (c) is used to determine coefficient b which is a function of the residents' perceived water quality index and whether the water body type is a lake or bay.

Equation (d) determines the constant term b which serves the function of making the change in property values equal to zero at the outermost limit of the area impacted by the project. Equation (e) represents the mathematical expression of the model's relationship where the percentage change in property values ($\Delta P\%_d$) is a function of both the perceived water quality changes by residents as predicted by experts (embodied in b_1) and the average distance from the water zone d ($1/DW_d$).

The period of project analysis was determined to be 34 years, 1976-2010; water resource experts estimate this to be the longest time period for a positive (with project) or negative (without project) change in water quality to occur either on Mirror or Shadow Lakes. With a limnologist's predictions of the status of the seven water quality parameters for each year, equations (a-e) are calculated, and the incremental, annual percentage change in property values is determined for different distances from the lakes.

To simplify the calculations, the impact area around the lakes was divided into separate distance-from-the-lake zones. Since Shadow Lake has ample public access, all residential property values throughout the City of Waupaca were assumed to be impacted by the project. Non-residential property was excluded (Lind, 1973). Ten distance-from-the-lake zones were constructed emanating out from Shadow Lake. For Mirror Lake only one distance-from-the-lake zone was constructed. As there is little public access, only lakefront owners were assumed to benefit from improvements in that lake's water quality.

The direct project benefits are calculated by multiplying the incremental percentage change in property values for each distance zone by the corresponding sum of property values in that zone; this is done for each year for both lakes. However, these benefits are spread over the entire life of the project, and to compare this stream of project benefits with the time stream of project costs, each must be reduced to a single number — their present value. The discount rate is the crucial parameter in this calculation. There are numerous, conflicting schools of thought regarding the appropriate discount rate. We used two rates to bracket this range: 7 1/8 percent and 15 percent, which reflect the rate suggested by the Water Resources Council to discount Federal projects and the opportunity cost of capital in the private sector as approximated by the prime lending rate, respectively.

The present values (1977 dollars) of project benefits and costs were used to determine the benefit-cost ratio for the project. If the ratio is greater than one, the present value of discounted project benefit exceeds that of discounted project costs and the project has met at least a minimum standard of economic efficiency. Discounted project costs were $439,872 and $469,650 for 7 1/8 percent and 15 percent discount rates, respectively. Discounted project benefits were $1,049,269 and $833, 958 for 7 1/8 percent and 15 percent discount rates, respectively. The two benefit-cost ratios generated in this study result from the sensitivity analysis performed with respect to the discount rates used. The corresponding benefit-cost ratios are 2.385 and 1.776 with 7 1/8 percent and 15 percent discount rates, respectively. These results indicate that regardless of the discount rate the project is justifiable using economic efficiency criteria.

Equity

Benefit-cost ratios address the issue of allocating scarce resources in an efficient manner. However, the preceding analysis ignores equity considerations regarding the distribution of benefits and costs. There are several equity considerations involved with the Mirror/Shadow Lakes Project.

We have assumed that the Federal, State, and local cost shares have been appropriately determined and we will only examine distribution of the local cost share. There are basically two approaches for subsidizing the satisfaction of public wants — the "ability to pay approach" and the "benefits received approach." The former is typically implemented through taxes on property and the latter by special assessment taxation in proportion to benefits received.

Local revenues for this project were raised by levying a 2-year 0.9 mill rate tax on all real property in the lake district (city). Each property owner then paid for a portion of the project according to his/her assessed property (e.g., approximately $90 for a $50,000 home). However, the well-being that each residential property owner enjoys as a result of the project does not vary in the same manner as the amount of taxes each had to pay. One reason for this discrepancy is that the increases in residential property values, because of a perceived improvement in water quality, diminishes as distance-from-the lake increases. An example of the discrepancy between the distribution of project benefits and costs can best be demonstrated by examining the Mirror Lake properties. The analysis reveals that 31 percent of the benefits accrued to these properties; however, only 5 percent of the costs were paid by these property owners.

If local financing was meant to be distributed on benefits received basis, then instead of a uniform mill rate being levied on all property owners, a special assessment based on a graduated rate could be implemented to reflect diminishing property value benefits for homes farther away from the lakes. Those homes that have lake frontage on Mirror Lake could be levied an even higher special assessment to reflect the exclusive benefits they enjoy from the improvements in that lake's water quality. And if financing was meant to be distributed on an ability-to-pay basis special consideration should still be given to the expected flow of benefits as the higher valued properties in this instance are also the ones to benefit the most since they are the ones located in the zones closer to the lakes.

SOCIOLOGICAL IMPACTS

Many public projects, especially large Federal projects, have been criticized as being insensitive to human needs. Decisions to undertake a project are often based on narrow economic criteria. While economic benefits are calculated to be greater than economic costs, the social costs to the residents and community are often greater than the social benefits (Dixon, 1978). The controversy surrounding such projects is the result of inadequate attention to social impacts.

Such a controversy did not develop before, during, or after the lake project in Waupaca; the social impacts analyzed were neutral or positive.

Citizen Participation

Since a city council can create and operate a lake district under Wisconsin law (Klessig, 1979), the Waupaca City Council could act without a petition from landowners and without extensive involvement by citizens. The city could also use its administrative staff to implement the project and supervise contractors.

The minimal citizen participation is shown by comparing attendance figures at annual meetings in Waupaca with those of a similar project in a small population rural setting at White Clay Lake. Only 2 percent of Waupaca residents attended a lake district annual meeting. In contrast, half of the White Clay Lake residents attended such meetings. In the rural area of White Clay Lake, without an incorporated local government, 14 percent attended as many as eight annual and special meetings (Klessig and Lovejoy, 1980).

Environmental Understanding

To determine the impact of the project on the knowledge level of citizens, a series of questions on lakes were asked of the Waupaca sample, the White Clay Lake property owners, and a statewide control group. Table 1 shows that Waupaca residents scored very close to the statewide average. They scored highest on a storm sewer question — one directly related to their project. Beyond that specific issue, there appeared to be little increase in knowledge about lakes. On the other hand, White Clay Lake residents generally scored substantially above the State average. This difference may reflect the greater participation by White Clay Lake citizens. Rural location and farm occupations may have also contributed to greater knowledge of the lake ecosystem.

Community Cohesion and Development

In many situations, a local community's involvement with large projects yields valuable experience in terms of personal familiarity with granting agencies, knowledge of technical and financial assistance, and assertiveness in dealing with bureaucrats. In other cases it yields frustration, bitterness, distrust of government and unwillingness to participate in future programs. Thirty-four percent of Waupaca residents felt the project experience would be useful to Waupaca in the future. Most of the remaining residents were not aware of the project.

There was little evidence that residents of Waupaca felt the project had damaged community development or community cohesion. At no time did the project open or reopen wounds between segments of the community. The lake project was not the type of development that pitted old against young, newcomers against traditional families, or developers against environmentalists. When asked whether the lake management project made their community a more or less desirable place to live, a majority felt that their community was a desirable place before, and the project had not affected that. Thirty-six percent felt the project had made Waupaca more desirable. No respondent felt the project had had a negative impact.

Table 1. — Educational impacts of lake projects in percent correct responses. Italicized words are correct answers.

	Waupaca	White Clay	Statewide
1. City and village storm drains can empty into nearby lakes without hurting the quality of lake water. Agree or ***disagree***?	73	80	69
2. The major cause of lake fish dying — or fish kills — in the winter months is that the water gets too cold for the fish to live. Agree or ***disagree***?	70	97	72
3. If farmers near lakes fertilize their fields by spreading manure only in the winter, the amount of pollutants running to the lakes would be reduced. Agree or ***disagree***?	45	71	49
4. Marshes around lakes act as a filter because they keep out material which would otherwise pollute lakes. Agree or ***disagree***?	58	86	60
5. The lakes would always remain clear, clean and fresh if there were no people around to cause pollution. Agree or ***disagree***?	41	31	39
	(N = 140)	(N = 35)	(N = 1,342)

Table 2. — Excellent or good ratings on agency water quality protection activities by respondents who were aware of the agency's activity (N).

	Statewide	Waupaca	White Clay Lake
U.S. Environmental Protection Agency	49% (416)	68% (34)	55% (31)
Wisconsin Department of Natural Resources	56% (820)	72% (82)	47% (32)
University of Wisconsin-Extension	74% (316)	86% (43)	74% (27)
U.S. Soil Conservation Service	60% (417)	84% (51)	75% (28)
Regional Planning Commission	42% (296)	78% (27)	42% (24)

The project appeared to be perceived as one of a number of activities that were important in keeping Waupaca desirable.

Alienation/Agency Image

Another common result of large projects is impersonal decisionmaking. Citizens often feel overwhelmed by bureaucratic processes. They feel helpless to cope with big government, big labor, or big business. Decisions always seem to be made by an anonymous person in a faraway city.

The Waupaca lake project did not increase alienation. Both the Federal and State programs are offered to local communities rather than carried out directly by the agency. Table 2 shows how Waupaca residents, White Clay residents, and a statewide control group rated five related agencies on water quality activities. In comparison to the statewide sample, Waupaca residents rated all agencies higher. Two-thirds or more felt the agencies were doing a good or excellent job. The University of Wisconsin Extension received the highest rating, 86 percent, and the lowest was 68 percent for EPA. The anti-government feeling was much stronger in the statewide sample which had not experienced a lake project.

More Tourists

Tourism was the one project-related concern evident in Waupaca. Out-of-town residents made up a substantial portion of lake users prior to the project. A majority of the Waupaca residents indicated that they would not favor any increase in tourists. Over 80 percent were not in favor of increases over 25 percent.

Tourists present a special dilemma for lake management programs. State and Federal assistance is premised on use of local lakes; if the general public can't use a lake, why should their tax dollars be invested there? Local citizens, on the other hand, are reluctant to invest their time and money to manage the lake if they might be crowded out by "rowdy outsiders."

Economic benefits of projects are often calculated in terms of increased use by tourists who stimulate the local economy with their purchases. Local businesses may promote a project for this reason. However, local property owners usually would rather not share their lake with any more users. Tourists increase density at local facilities, recreational and commercial; this increased density may negatively affect local citizens and could promote community conflict.

SUMMARY

The Waupaca Lake District carried out a major project of storm sewer diversion, alum treatment, and aeration over a period of 5 years without any major setbacks or negative impacts. In economic terms, the project is generating more benefits than the $430,000 invested. While those near the lake might have been expected to pay a higher share of the local costs, the uniform property tax was modest and was simplest to collect with a uniform mill rate.

Social benefits have been positive and modest with the single exception that the project could become a liability for many residents, if tourism significantly increased crowding. Because the city council conducted the affairs of the lake district, the project provided little experience in self-governance for citizens or education in aquatic ecosystems. The residents liked Waupaca before the project and the lake project maintained that image. The image that residents held of government agencies was substantially improved during the course of the project. Most significantly, the project has not caused the community to suffer social costs, especially those which cut lasting divisions into the community structure. The Waupaca lake project went very smoothly; it enhanced the lake and maintained a functional social structure.

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LAKE MANAGEMENT AND COST-BENEFIT ANALYSIS IN ONTARIO

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ABSTRACT

Cost-benefit analysis is a highly developed, formal, economic methodology for evaluating the benefits and costs of a wide range of activities. It offers a systematic means for comparing lake management options so that an optimal mix of uses (e.g., recreation, waste disposal, potable water supply) can be identified. Such an approach would seem to be especially attractive during a time in which economic problems appear paramount and protection of the environment should be secured at the least possible cost. It is significant therefore, that the regulatory authorities in Ontario make little use of cost-benefit analysis for lake management purposes. This paper examines the reasons for this. Considerable emphasis is placed on the institutional framework for lake management in Ontario, and a liberal use is made of real life examples. The paper concludes with some recommendations on the role of cost-benefit analysis for lake management in Ontario, and by extension, in other similar jurisdictions.

INTRODUCTION

Ontario is Canada's third largest Province with a total area in excess of 1,036,001 square kilometers (400,000 square miles). More than 10 percent of the Province is covered by water. This is divided about equally between Canada's share of the Great Lakes and the rest of Ontario's lakes and rivers. In this age of quantification, there has been no official enumeration of Ontario's lakes. While they are not literally countless, they remain uncounted.

The abundance of Ontario's inland lakes has two important and contrary implications for water quality management in the Province. Because they are so numerous and frequently so large, the likelihood of serious, widespread contamination is reduced. However, the extent to which resources must be stretched to monitor and regulate activities which could damage the lakes makes it difficult for the regulatory authorities to perform their role effectively.

It should also be remembered that while the average population density of the Province is little more than 3/sq. mile, over 80 percent of the population live in urban centers. Most of the population is located in the southern part of the Province, which is the industrial heartland of Canada accounting for about 40 percent of the gross national product. This combination of high economic activity, with its related environmental impacts, and localized concentrations of a population accustomed to a wide variety of outdoor recreation, places a considerable burden on many of the Province's more accessible lakes.

Supplementing these domestic sources of actual and potential adverse impacts on Ontario's lakes are those for which people outside the Province are responsible. Each year some 20 million Americans visit Ontario, many for recreational purposes. Furthermore, sulfuric and nitrous oxides transported through the air internationally may be having irreversible impacts on numerous lakes in the Province.

All of these circumstances taken together present a formidable challenge to rational and effective lake management in Ontario. It is a challenge which has been met, though by no means with complete success, primarily at the Provincial level of government rather than Federal or municipal. Accordingly, this paper focuses on the approach to lake management and especially benefit assessment, taken by the Provincial authorities. However, in the case of the Lake Simcoe-Couchiching basin study which is discussed at some length, the role of the municipal authorities is readily acknowledged.

The next section outlines lake management policies and implementation procedures in Ontario, and insofar as the treatment is specific, it deals with water quality management in lakes other than the Great Lakes. There then follows an account of a recent attempt to develop an environmental strategy for Lake Simcoe-Couchiching, southern Ontario's largest body of water after the Great Lakes. The limited consideration given to the benefits from improved water quality in this otherwise comprehensive study is especially noteworthy.

Finally, the paper considers, from a somewhat critical standpoint, the role that cost-benefit analysis might play in Ontario's lake management and examines the reasons why the Province has made relatively little use of this evaluation technique.

ONTARIO'S APPROACH TO LAKE MANAGEMENT

Though several Provincial Ministries have a role to play in lake management in Ontario, the primary responsibility for water quality management rests with the Ontario Ministry of the Environment. The goal of the Ministry with respect to surface water quality is: "to ensure that the surface waters of the Province are of a quality which is satisfactory for aquatic life and recreation" (Ontario Minist. Environ. 1978).

It is believed by the Ministry of the Environment that "water which meets the water quality criteria (designated as Provincial Water Quality Objectives) for aquatic life and recreation will be suitable for most other beneficial uses, such as drinking water and agriculture"(Ontario Minist. Environ. 1978).

A major policy implication of this general goal is that the use of stream classification, where specific rivers and lakes in the province are designated for various and different uses, is not permitted since all surface water must be suitable for all uses. In fact, this goal has not been achieved and implicitly stream classification is practiced in Ontario, at least in relation to setting timetables for compliance with effluent discharge objectives.

The Ministry of the Environment has set Water Quality Objectives for lakes and rivers which, if satisfied, will fulfill the Ministry's water quality management goal. These Objectives are both quantitative, e.g., expressed as concentrations, and qualitative where conditions of the receiving waters are declared unacceptable. The Objectives were not established to balance costs and benefits. Benefits are assumed, and cost considerations only enter in cases where it is acknowledged that the Objectives cannot be met owing to the accumulation of past discharges.

The stage in the regulatory process where benefits (and costs) do play some role is in the Ministry's compliance programs for point source discharges. (To date, the Ministry has done little to regulate nonpoint sources of wastewater contaminants.) For industrial sources the Ministry works out discharge objectives on a case by case basis. Compliance schedules are negotiated and a company may receive a "program approval." Providing the terms of the approval are not contravened, the company cannot be prosecuted for pollution until the time period of the approval has run out. Failure to comply with the terms of a program approval is not in itself an offense. Partly because program approvals have not achieved abatement objectives, Control Orders and Requirements and Directives have been increasingly emphasized. These are legally enforceable statements requiring companies to undertake studies and to control their waste discharges. Non-compliance with either of these regulatory instruments is punishable with a fine.

Note that Ontario has no effluent discharge standards for water pollution. In effect, a Control Order establishes a source specific standard but the issue of a Control Order is entirely at the discretion of the Ministry of the Environment, as is an amendment to a Control Order. Likewise, "Certificates of Approval" which are licenses required by anyone wishing to operate a potential source of pollution, are issued by the Ministry and are tailored to the conditions of the receiving waters.

The other major category of point source discharges into Ontario's lakes and rivers is municipal sewage treatment plants. Typically these have been built by the Ministry of the Environment (and its predecessor, the Ontario Water Resources Commission) and in most cases, turned over to the municipalities for operation once satisfactory performance has been achieved. This close involvement of the Ministry in the construction and operation of these plants has facilitated somewhat more effective control than is the case for many industrial sources. Nevertheless, problems can arise, especially when the Ministry seeks improvements in the performance of municipal sewage treatment plants beyond the original design capability, since the municipalities may be reluctant to incur the associated increase in costs.

This brief description of Ontario's approach to achieving and maintaining a satisfactory level of water quality in lakes and rivers should not obscure the fact that it is all part of a comprehensive regulatory framework. In addition to surface quality the Ministry of Environment has policies, objectives, and implementation procedures for surface water quantity management, which limit water withdrawals, and for groundwater quality and quantity management. This activity is supported by an extensive and sophisticated research capability. Moreover, the Ministry, working through its regional and district offices as well as the head office, liaises with staff in other Ministries and government agencies whose responsibilities also impinge on lake management in the Province. Important among these are the Ministry of Natural Resources, which regulates the exploitation of such resources as wildlife and forests, and the municipally-based Conservation Authorities, with responsibility for land-use and development activities as they affect flood-plain areas and the total watershed.

Notwithstanding all this planning and regulatory activity, and the enormous production of studies and data, little explicit consideration is given to the benefits of lake management in Ontario. This is further illustrated in the development of an environmental strategy for the Lake Simcoe-Couchiching Basin described in the next section.

THE LAKE SIMCOE—COUCHICHING BASIN ENVIRONMENTAL STRATEGY: THE ROLE OF BENEFIT ASSESSMENT

Lakes Simcoe and Couchiching have a combined area of some 777 square kilometers (300 square miles) draining a land area of 2,434 square kilometers (940 square miles) (see Figure 1). Proximity to large centers of population and outstanding natural features have led to increasing use of the lakes for swimming, fishing (sports and commercial), boating, water supply (cottages and small townships), and waste disposal (municipalities and cottages). A 4-year study of the lakes culminated in a report published in 1975 (Ontario Minist. Environ. 1975) which concluded that local problems aside, the general water quality of the lakes was good, but problems were emerging. Chief concerns centered around increasing algal growths and changing fishing success. These were related to excess nutrient material, particularly phosphorus, being discharged into the lakes.

As a result of public meetings and the further accumulation of data by Provincial Ministries, two committees were formed: a Report Committee and a Steering Committee. The Report Committee, consisting of staff from Provincial Ministries and municipalities, was directed by the Provincial Cabinet Committee on Resources Development (CCRD) to: (1) Assess the types and magnitudes of environmental problems in the Lake Simcoe-Couchiching area; (2) identify the causes of these problems; and (3) propose a strategy for dealing with the problems.

The proposals of the Report Committee were adopted by the Steering Committee whose members were drawn primarily from the local municipalities. After receiving the report CCRD accepted all of the recommendations except that referring to the critical issue of phosphorus loadings to the lakes.

The Report Committee estimated the total phosphorus loading to the lakes in 1979 to be 103 tons, broken down by sources as shown in Table 1.

The Report Committee also identified other environmental problems in the watershed: Important marsh and wildlife areas are being encroached upon, forested areas are being diminished, poorly managed mining activities are threatening ground waters, and sensitive ecological areas are subjected to stress. All of these factors affect the basin environment and the recommended environmental strategy addresses them all. However, for the purposes of this paper it is sufficient to focus on the evaluation of benefits in relation to a reduction in phosphorus discharged to the lakes. This is also consistent with the emphasis given to this matter in the Simcoe-Couchiching report and in CCRD's response.

Three alternative environmental development strategies were considered by the Report and Steering Committees:

1. Maintain existing environmental quality (i.e., maintain present water quality, fishery, and general basin environment);
2. Improve environmental quality;
3. Allow environmental deterioration.

This third option was rejected by both committees on the grounds that it is contrary to the Provincial policy on surface water quality management and to the desires of the local community. No attempt was made to identify and evaluate the benefits that would be foregone if this alternative were adopted. It was assumed implicitly that these would outweigh any savings in costs that such a strategy would allow.

According to the report, maintaining existing environmental quality in the face of local population growth and the necessary use of the lake for recreation requires deliberate actions to limit total phosphorus loadings to the current 103 tons/year. Any combination of schemes to control the individual sources of phosphorus which have the effect of limiting the phosphorus loading to this level was deemed a potentially acceptable strategy.

The benefits expected from maintaining existing water quality were believed to stem from "an abundance of warm-water species in Lake Simcoe and a precarious cold-water fishery." (Lake Simcoe-Couch. Rep. Comm. 1979). This in turn would benefit the tourism industry which "relies on the quality of the recreational experience which relates to good water quality and fisheries." (Lake Simcoe-Couch. Rep. Comm. 1979).

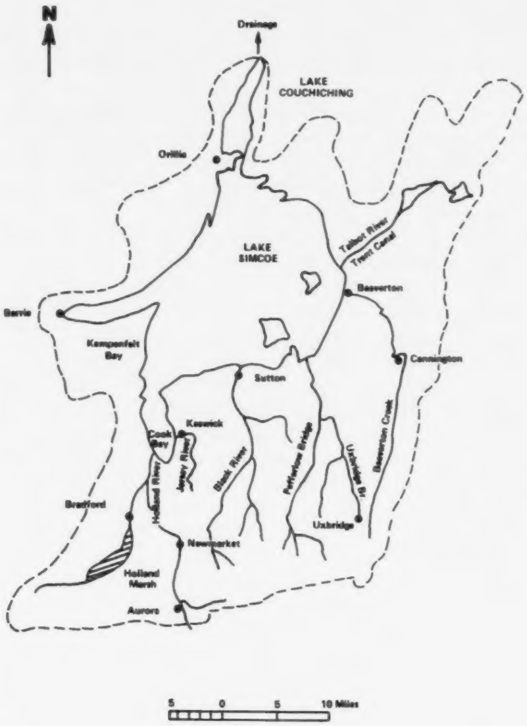

Figure 1. — Map of Lake Simcoe-Couchiching drainage basin.

Table 1. — Estimated phosphorus loadings into Lakes Simcoe-Couchiching, 1979.

	Tons/Year* Phosphorus
The field leakage — cottages	3
Precipitation	21
Rivers under natural watershed conditions	26
Urban storm wastes	9
Agriculture and other land-use disturbances	22
Sewage treatment plant effluents (with P removed to 1 mg/l)	22
	103

* 1 ton (metric) = 2,000 lbs.
Source: Lake Simcoe-Couchiching Basin Environmental Strategy, 1979, p. 4.

An improvement in water quality, the last option to be considered would result in "the elimination of scums on the lakes, decreased weed growth, improved water clarity in some areas, and a more stable fishery with healthy, self-reproducing populations of whitefish and lake trout." (Lake Simcoe-Couch. Rep. Comm. 1979). (Under a strategy of maintaining water quality the whitefish population is expected to be extinct in 20 years and the lake trout population might be maintained through stocking.) As for the phosphorus loading required by this strategy of improvement it would have to be reduced progressively to 75 tons/year to achieve a self-reproducing cold water fishery. (Lake Simcoe-Couch. Rep. Comm. 1979). Again, the study did not give any detailed consideration to the value of the benefits that might be derived from such a strategy. No more on benefits that has been given in this brief summary of the study was produced for comparison with the roughly estimated strategy of:

1. Maintaining existing water quality: $3,000,000/yr (1979 Canadian dollars) with a basin population increasing from 190,000 to 300,000.

2. Improving water quality — $4 to $5,000,000/year with no increase in basin population.

The Report Committee recommended a total phosphorus loading objective of 103 tons/year. This was reduced somewhat by the Steering Committee to 95 tons/year to allow for some uncertainty in the performance of the proposed control measures. Subsequently, the Cabinet Committee on Resources Development reduced the phosphorus objective to 87 tons/year. What is especially significant here is that this was based on a belief that "the value of the cold water fishery in terms of tourism and the economy, as well as an indicator of the ecological health of an important water resource, warrants a major effort to restore water quality to a level which will support such a fishery. . . .Improved water quality, and the resulting reduction in slime and aquatic weeds, will make the basin a more attractive recreation and tourism destination and could increase property values" (Cab. Comm. Resour. Dev. 1979).

This objective of 87 tons/year substantially exceeds the 75 tons/year stated by the Report Committee to be necessary for a self-reproducing cold-water fishery. Yet apparently, the CCRD had no new evidence on which to base its belief that substantial benefits from a thriving cold-water fishery would ensue if this revised objective is achieved. To give such weight to anticipated benefits when the supporting documentation is so obviously lacking underlines the importance of improving benefit estimation in developing lake management strategies. It is the potential of cost-benefit analysis in this regard that is discussed in the next section.

COST—BENEFIT ANALYSIS: BENEFIT ESTIMATION AND LAKE MANAGEMENT

Cost-benefit analysis is a highly developed, formal, economic methodology for evaluating the benefits and costs of a wide range of activities. It offers a systematic means for comparing lake management options so that an optimal mix of uses (e.g. recreation, waste disposal, water supply) can be identified.

Such an approach would seem to be especially attractive during a time in which economic problems appear paramount and protection of the environment should be secured at the least possible cost. It is significant therefore, that the regulatory authorities in Ontario make little use of cost-benefit analysis for lake management purposes. In particular, the estimation and evaluation of benefits within the cost-benefit framework is seldom practiced in Ontario.* The discussion which follows will examine possible reasons for this. It will also consider whether the use of such an approach might improve lake management in Ontario and, by extension, in other similar jurisdictions.

The first possibility is that the benefits from lake management are already adequately accounted for in the regulatory process. This does not seem to be the case either in the establishment of Ontario's Surface Water Quality Goal or in the setting of the Provincial Water Quality Objectives. When it comes to the actual process of regulation, which involves a considerable degree of informal negotiation between the regulators and those responsible for waste discharges, the picture is less clear. Nevertheless, the development of the Lake Simcoe—Couchiching management strategy, which stands out as a relatively comprehensive and systematic approach to lake management, underlines the casual way in which benefits are addressed. It does not appear, therefore, that adequate consideration is already given to benefits in Ontario's approach to lake management.

A second possible reason for not using cost-benefit analysis in Ontario's lake management is that the approach is not well understood by Ontario's regulatory authorities. The Ontario Ministry of the Environment, like many other environmental agencies, is staffed principally by people with backgrounds in engineering and the natural sciences, who characteristically approach environmental management differently than an economist. The notion of optimizing across a range of environmental, social, and economic objectives is alien to them and this tends to weaken the appeal of an approach such as cost-benefit analysis.

The reasons given so far question the adequacy of the existing approach, and the orientation of key personnel within the regulatory agencies. The possibility that deficiencies in cost-benefit analysis might account for its lack of use must now be examined.

Cost-benefit analysis in any application is essentially a process of market simulation (Mishan, 1976). The analysis attempts to identify the potential gainers and losers from a proposed project, program, or policy and to estimate the maximum sum of money the gainers would be willing to pay for the benefits and the minimum that the loser would require as compensation for incurring the losses. Only if the benefits exceed the costs, according to these definitions, does the proposal being analyzed pass the cost-benefit test.

The technical problems involved in conducting a cost-benefit analysis are challenging even to the most well-trained and experienced economist. The information requirements, alone, may be too demanding, especially on the benefits side, to allow the use of this approach for lake management. But even if these difficulties can be overcome there are other problems with the approach.

First of all, the proper description of gainers and losers may be ambiguous. In the case of lake improvements requiring pollution abatement the gainers might be those who would benefit from improvements and the losers those who will have to incur costs to abate pollution. Alternatively, the same project could be looked at from the viewpoint of maintaining the existing level of water quality. In that case the gainers would be those not having to further control their pollution and the losers would be those having to forego the benefits from enhanced water quality. A cost-benefit analysis could be conducted from either perspective and the results could well be different.

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* Cost-effectiveness, a subcomponent of cost-benefit analysis is sometimes used to determine the least costly means of achieving some prescribed objective. Furthermore, the Ontario Ministry of the Environment has recently (July 1980) funded studies of the damages due to acid precipitation.

Typically, economists presume that the status quo is an acceptable point from which to start and so the gainers would be those who would benefit from the improved water quality. By implication, this confers a right to pollute on existing waste dischargers and this may be unacceptable to the regulators and to the public at large.

Another way in which the status quo is often given normative significance in cost-benefit analysis is with respect to the distribution of incomes and wealth. Estimates of willingness to pay for benefits and compensation required for costs depend upon people's economic situation. A change in this will change the estimates. Again it may be unacceptable to the regulators and to the public for decisions about the proper use of the publicly shared environment to reflect the distribution of private incomes and wealth. Yet this is built into cost-benefit analysis and can only be amended through arbitrary adjustments in the benefit and cost estimates.

Some economists argue that it is appropriate for cost-benefit analysis to deal only with "economic efficiency" (Mishan, 1976). Considerations of equity or fairness, such as those alluded to here, should be addressed in the political arena. Insofar as lake management poses problems of equity, both among contemporaries and across generations, this further justifies limiting the role of cost-benefit analysis in lake management. Moreover, it is questionable whether efficiency (getting the most from the least) and equity (sharing the benefits and costs fairly) can be separated in this way when the measures of costs and benefits depend upon the distribution of incomes and wealth.

Other issues which pose difficulties for cost-benefit analysis in lake management relate to the scope of the analysis both temporally and spatially. How should benefits and costs expected in the distant future be compared with those likely to be incurred in the near term? Should the gainers and losers include people from beyond the jurisdiction responsible for lake management? (This is particularly important in Ontario where many of those who benefit from improvements in lake quality come from other Provinces and countries.) But these are questions that any rational approach to lake management must confront. Though they may be inadequate, answers to them are provided within the benefit-cost framework. Intertemporal comparisons of benefits and costs are made using a discount rate which has the effect of giving more weight to benefits and costs the sooner they are expected. The scope of a benefit-cost study in terms of who is included typically reflects the extent of the jurisdiction within which the analysis is being done.

Finally, cost-benefit analysis may be regarded with some skepticism by those who believe that decisions on lake management should reflect concerns that override those of human beings alone. Responsibility for the protection of the environment goes beyond questions of its optimal use. Whether or not this perspective is compatible with the cost-benefit framework, the obvious anthropocentric bias of cost-benefit analysis has no doubt deterred some people from taking it more seriously.

CONCLUSION

This paper has shown that systematic benefit estimation plays a minor role in lake management in Ontario. It has also suggested several reasons to explain why cost-benefit analysis has not been used more extensively, especially for evaluating benefits. The question remains whether, despite all its real and perceived shortcomings, cost-benefit analysis could usefully contribute to the formulation of lake management strategies and programs. The greatest danger in this regard lies in the possibility that poorly conducted cost-benefit analyses, from which numerous important considerations are omitted, should come to supplant the sort of participative approach being developed in studies such as that for the Lake Simcoe-Couchiching Basin. While much may be inadequate about that process, the opportunity for a wide range of affected parties to interact, provide information, and learn is impressive. Yet, a greater emphasis on benefits, however approached, might improve the process considerably. At the very least, the categories of benefits (e.g., water supply, recreation, habitat, commercial fishing) expected from lake management could be specified, case by case. A modest effort might provide quantitative estimates of the relationship between various levels of lake water quality and some of these benefits measured in natural units (e.g., gallons of potable water, recreation days, acres of habitat, tons of catch). Ascribing dollar values to these benefits, so that they may be aggregated and compared with the costs of lake management, is the final step in cost-benefit analysis and no doubt the most treacherous one. But if it is not taken some other possibly inferior means must be found to evaluate lake management options.

Cost-benefit analysis is one way to increase the attention given to benefits and costs in lake management; it makes explicit issues that have to be dealt with one way or another in any case. Providing the estimates of benefits and costs are properly presented and understood, they could enhance the planning and regulation of lakes in Ontario and elsewhere.

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THE LEMAN COMMISSION

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ABSTRACT

The Commission Internationale pour la Protection des eaux du lac Leman et du Rhone contre la pollution was officially established in 1960, originating from an informal Franco-Swiss institution founded 10 years before. Study of the sanitary and trophic status of a lake is followed by a technical subcommission involving several laboratories of both countries. The studies concern the evolution over time and space of the different components of the whole ecosystem: The lake (582 km^2) and its drainage basin (7,390 km^2). The studies are planned on a 5-year basis and distributed to the different laboratories according to their specialization (chemistry, biology, microbiology). The results are published in annual reports and constitute the basis for the Commission's recommendations. The main practical objective was to diminish eutrophication by domestic and industrial sewage treatment, including phosphorus elimination. More recently the Commission has been faced with problems of mercury and PCB pollution.

INTRODUCTION

With 8,582 km,^{3} the Leman —Lake Geneva— is the greatest lake and the largest freshwater reserve in western Europe. The lake and its drainage basin are shared by France and Switzerland. An international commission has been founded to solve pollution problems through a joint effort of everybody concerned.

THE LAKE AND ITS DRAINAGE BASIN

The Leman includes two sub-basins: The Grand Lac upstream, and the Petit Lac downstream (Figure 1). Table 1 gives their main physical characteristics. The Petit Lac looks more like an enlarged river than a lake; it is more realistic to express its mean residence time versus depth:

4 to 5 years from the surface to 50 meters deep
10 years from 50 meters to 200 meters
20 years from 200 meters to the bottom.

The area distribution between France and Switzerland is given in Table 2.

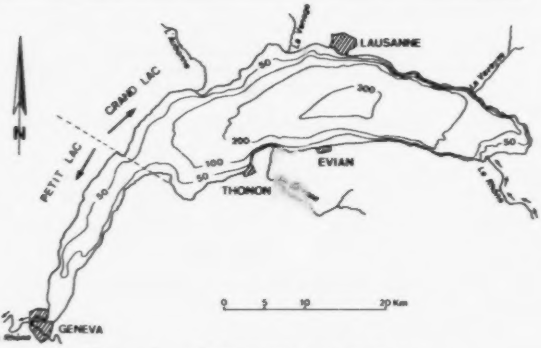

Figure 1. — Bathymetric map of the Leman.

The Leman receives water from a surface area of 7,390 km^2, 80 percent of which lies in Switzerland (Figure 2). With 176 $m^3 S^{-1}$, the Rhone River is the main tributary and represents 75 percent of the water input. Its drainage basin, totally under Swiss control, culminates at an altitude of 4,638 meters and is 5,220 km^2 wide; 16 percent of the surface area is covered with glaciers. The Dranse River is the second largest tributary, draining 535 km^2 in France for a contribution of 20 $m^3 S^{-1}$. Urban, industrial, and agricultural activities are concentrated in the plains and dispersed in the mountains. The principal activity seen in the mountains is tourism.

COMMISSION ORIGINS

During a meeting in Lyon, France, in the spring of 1950, the Association of Rhodanians (Union Generale des Rhodaniens) pointed out the disastrous conse-

Table 1. — Main physical characteristics of the Leman.

	Grand lac	Petit lac	Leman
Surface area (km^2)	503.5	78.8	582.3
Surface area (%)	86.5	13.5	100.0
Volume (km^3)	85.69	3.23	88.92
Volume (%)	96.4	3.6	100.0
Maximum depth (m)	309.7	76.5	309.7
Mean depth (m)	172.4	37.8	152.7

Table 2. — Distribution between France and Switzerland.

Swiss Canton or French Department	Surface area	
Vaud, Switzerland	293.98 km^2	
Geneve, Switzerland	36.35 km^2	= 340.83 km^2
Valais, Switzerland	10.50 km^2	
Haute-Savoie, France	241.47 km^2	

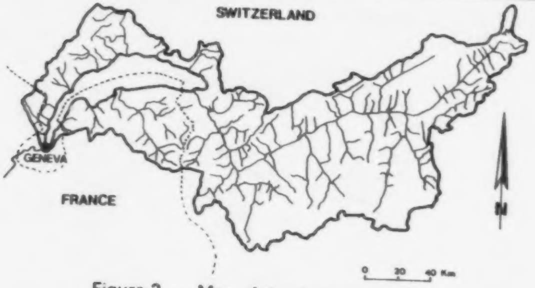

Figure 2. — Map of the drainage basin.

quences for man and his environment of dumping sewage into the Rhone. Dr. Messerli from Lausanne, Switzerland, was asked to establish an informal commission to study sewage pollution and to try to find a solution through coordinated action of the concerned populations. This commission consisted of many experts generally connected with French or Swiss authorities.

After a few years devoted to standardizing analytical methods and elaborating sampling programs, the first systematic and coordinated survey began in 1957. But because of its informal status, this commission was limited to observation and research without any practical application. Therefore, on November 9, 1960, Swiss and French authorities decided to recognize the commission officially, as the Commission Internationale pour la protection des eaux du lac Leman et du Rhone contre la pollution. A Franco-Swiss agreement became effective November 1, 1963.

THE FRANCO-SWISS CONVENTION

Established between Switzerland's Conseil Federal and the French government to coordinate their efforts to protect Lake Geneva against pollution, the Convention extends the Commission's jurisdiction to the Swiss border at the lake outlet and includes all superficial and deep waters. It establishes the Commission's four main objectives:

1. To organize and monitor all research aimed at determining the nature, extent, and origin of pollution and to use the results.
2. To recommend governmental measures to cure today's pollution and to prevent future pollution.
3. To prepare the basis for establishing international regulations in the case of incompatibility between legislation of the respective governments.
4. To investigate all questions concerning water pollution.

The result is that only strictly-applied research is developed and entrusted to a sub-commission (the Sous-Commission Technique). Occasional working groups may be constituted for solving specific problems. Finally, the Convention limits the Commission's intervention to recommendations, all decisions being made by the governments.

THE COMMISSION'S MEMBERSHIP

The Commission consists of two delegations; the French delegation is led by Mr. J. Leclerc from the Foreign Office in Paris and is composed of eminent officials from concerned governmental authorities; the Swiss delegation is led by Mr. Pedroli, Director of the Federal Office for Environmental Protection in Berne. The delegation leaders alternate in presiding at commission meetings. The Commission is organized as shown in Figure 3.

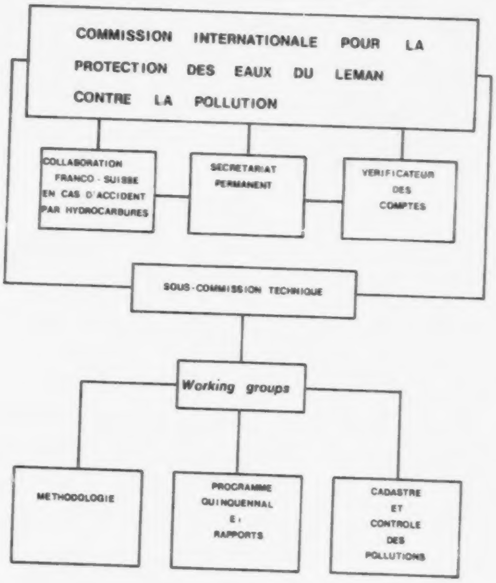
COMMISSION INTERNATIONALE POUR LA PROTECTION DES EAUX DU LEMAN CONTRE LA POLLUTION
COLLABORATION FRANCO - SUISSE EN CAS D'ACCIDENT PAR HYDROCARBURES
SECRETARIAT PERMANENT
VERIFICATEUR DES COMPTES
SOUS-COMMISSION TECHNIQUE
Working groups
METHODOLOGIE
PROGRAMME QUINQUENNAL E: RAPPORTS
CADASTRE ET CONTROLE DES POLLUTIONS

Figure 3. — Organization of the Leman Commission.

The S.C.T. (Technical Subcommission)

The Sous-Commission Technique is the Commission's executive body. It plans research on a 5-year basis and elaborates recommendations. It includes experts in several disciplines: Physicians, engineers, biologists, microbiologists, chemists, etc., who are selected by the delegation leaders. It is presided over alternately by a French or a Swiss representative. The Sous-Commission Technique is also divided in two delegations, functions according to internal regulation, and has its own administrative board.

The Secretariat Permanent (Permanent Secretariat)

This technical and scientific secretariat assists the Commission in everything concerning data treatment, annual report preparation, public information, etc.

Franco-Swiss collaboration on oil pollution

The Collaboration Franco-Suisse en cas d'accident par les hydrocarbures is totally independent of the Sous-Commission Technique and is directly attached to the Commission. It organizes and coordinates action taken against oil pollution. Its existence and its effectiveness result from a specific agreement, dated November 18, 1977, which solves problems concerning crossing the border on land, lake, and air.

Table 3. — Program of the purification plants construction.

	Sewage treatment plants							
	Number				Treatment capacity			
LEMAN	1968	1973	1978	1979	1968	1973	1978	1979
Valais	9	18	36	38	21,100	316,190	762,845	770,645
Vaud	8	35	58	57	262,000	507,000	666,630	668,200
Haute-Savoie	1	4	6	8	3,000	109,000	127,000	129,400
Ain	1	3	4	4	500	29,500	33,000	33,000
Geneve	3	3	3	3	4,900	5,050	6,950	6,950
Total	22	63	107	110	291,500	966,740	1,596,425	1,608,195
With phosphate elimination	-	-	48	45	-	-	1,448,760	1,449,730
RHONE'S BASIN								
Haute-Savoie	5	11	15	19	11,200	39,000	119,150	133,950
Ain	-	2	3	3	-	18,500	20,000	20,000
Geneve	4	14	13	13	400,000	451,000	460,505	460,505
Total	9	27	31	35	411,200	508,500	599,655	614,455
General total	31	90	138	145	702,700	1,475,240	2,196,080	2,222,650

THE COMMISSION'S AIMS

Lake Geneva's water quality must be appropriate for drinking water, bathing, and salmonids. The definition of water quality is based on European Economic Community parameters. According to the Commission's recommendations, many dispositions have been taken in connection with:

Domestic sewage: Table 3 gives the program of treatment plant construction realized since 1968. On January 1, 1979, the rate of collected populations was 73 percent in Lake Geneva's watershed and 56 percent in the downstream watershed; 45 of 110 treatment plants are currently eliminating phosphorus. Treatment efficiency is generally inspected once a year, 50 percent of the plants being checked four times. This frequency of inspection will be generalized; analysis will concern a 24-hour sample. BOD^5 elimination is better (<20 mgl^{-1}) than COD elimination and much better than phosphorus elimination. During 1978, only five plants serving 400,000 persons each, were able to reach the 1 mgPl^{-1} limit. The effluents of 20 plants contained between 1 and 2 mg. The situation is now improving, thanks to the Commission's encouraging work towards better phosphorus elimination. Recently, the Commission proposed establishing an international fund to buy reagents necessary for phosphorus elimination.

Phosphorus in detergents: The Commission recommends reducing or eliminating phosphate detergents. In the meantime, it recommends prohibiting or limiting television publicity for these products.

Nutrient nonpoint sources: Investigations are conducted to estimate the agricultural and the natural contributions to the nutrient budget. The Commission also recommends a better focus on pollution from touristic and commercial navigation through technical and legislative measures relating to boats and harbors.

Industrial pollution: During 1970-1972, investigations conducted on the Rhone demonstrated that 10 to 15 kg of mercury were introduced daily into the lake's ecosystem. Public opinion reflected significant agitation, exacerbated by mass media and by the Minamata affair. Necessary measures were rapidly taken, and in 1978 new investigations indicated a reduction to 1 to 3 kg of mercury per day. During the next 5 years heavy metals (lead, chromium, cadmium) and PCB will be intensively surveyed in sediments and fish populations.

CONCLUSIONS

Although the realizations appear to be slow, it must be remembered that the Commission is not invested with any supranational authority and that its direct action is limited to simple recommendations. It is at least a consolation to see they are not ineffective.

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STRUCTURE, AIMS AND ACTIVITIES OF THE INTERNATIONAL ALPINE COMMISSIONS IN EUROPE

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ABSTRACT

Structure, aims, and activities of the International Alpine Commissions in Europe are discussed. Particular emphasis is given to the constitution of the commissions and the most important problems concerning the common water of the five countries.

The boundaries between countries result from historical events, and, therefore, they do not coincide with the boundaries of the watersheds. As a consequence, several lakes and rivers mark the boundaries between countries or cross them. To effectively manage these water resources and protect them against pollution, the governments of the countries concerned must agree upon common rules and actions concerning this problem. On this basis, over the last two decades in the Alpine region of Europe several conventions between two or more governments have been ratified. These conventions entrust the protection of the surface and ground waters to international commissions. This important problem was exhaustively discussed at the OECD Seminaire sur la pollution transfrontiére dans les bassins hydrographiques internationaux (June 6-10, 1977).

The Alpine International Commissions for water protection are:

1. Lake Constance, created in 1959 and signed the following year by the governments of Austria, Baden-Waerttemberg, Bavaria, and Switzerland.
2. Lake Geneva, officially established in 1960 and ratified by the French and Swiss governments in 1963.
3. Lake Maggiore and Lake Lugano, created and signed in 1960 by the governments of Italy and Switzerland (and again in 1972 for subsequent water courses and the ground waters of their watersheds).

The structures of these commissions are very similar. Each has a president and is composed of delegates from member governments and a limited number of high officers of those governments. The presidency changes after defined periods. As a rule, the commissions meet at least once a year.

The most important duties of the commissions are the following:

1. To examine problems concerning water pollution and to judge proposals on studies of these waters as well as on the actions to be taken to reduce the pollution level;
2. To illustrate to the governments the water problems and propose actions for protecting them against pollution;
3. To establish financial plans to support the studies sanctioned by the commission.
4. To prepare the basis for establishing international regulations in the case of incompatibility between the legislation of the respective governments (as in the case of Lake Geneva).

As consultant agencies for the governments, the commissions cannot decide on rules and actions connected with environmental protection.

Technical and scientific sub-commissions serve as official consultants to commissions; their number is also limited. The sub-commissions have the following duties:

1. To study the scientific and technical problems proposed by the commission and examine the research carried out by other organizations;
2. To elaborate on the research program to be submitted to the commission;

3. To prepare the reports on the research sanctioned by the commission.

(The sub-comissions have working groups for studying special problems; external experts may be invited to collaborate with these groups).

As an example of the sub-commission activity, some of the most important problems investigated by the sub-commission are described here.

For the Italian-Swiss sub-commission these are:

1. Pluriannual researches on the trophic evolution of Lake Maggiore and Lake Lugano and the evaluation of the nutrient load (nitrogen and phosphorus compounds) from the watershed of these lakes;

2. Studies on the microbiology of Lake Maggiore waters and, particularly, the coastal area;

3. Studies for unifying criteria and methods to control the sanitary conditions of the swimming waters of Lake Maggiore and Lake Lugano. These studies have been carried out on the basis of a directive of the European Communities Council (December 8, 1975).

4. Comparison between plans for protecting and ameliorating the quality of the Swiss-Italian water bodies;

5. Elaboration of a unified plan for the alarm and intervention in the case of an accident produced by the release of hydrocarbons or other noxious substances in the Swiss-Italian water bodies.

In Lake Constance some important working groups are: Lake water research; river control; sediment investigations; oil, heavy metals, and organic allochthonous compounds (and other scientific problems); investment program for treatment plants; collaboration on oil pollution prevention and other technical problems. There are no activities on sanitary problems.

Some of the most important programs of the Lake Geneva commission are: Franco-Swiss collaboration on oil pollution protection; domestic sewage; phosphorus in detergents; nutrient nonpoint sources; and industrial pollution.

The sub-commissions meet four to five times per year. Their working methods are: (1) A problem is identified and discussed by the sub-commission experts; (2) a proposal for its solution is worked out and is given to the commission to judge. The financial support for major projects is guaranteed by the member governments.

At the sessions of the different working groups a regular reporting on the progress of the investigations takes place. The results are summarized in final reports and, as a rule, published. For the publications, special redaction committees exist. The publications are available from:

Lake Geneva Commission, Secretary at Lousanne Chailly (Dr R. Monond); Lake Maggiore and Lake Lugano Commissions, Secretary at Locarno-Casa Rusca (Dr. A. Rima); Lake Constance Commission, Ministries for Environmental Protection of the collaborating countries (Bern, Munich, Stuttgart, Vienna.)

INSTITUTIONAL ARRANGEMENTS FOR SHORELAND PROTECTION AND LAKE MANAGEMENT IN WISCONSIN

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ABSTRACT

Increasing development brings problems to many lakes. Proper shoreland development can be a key factor in avoiding lake problems. Where problems already exist, certain management practices often can prevent further degradation or repair past damage. Institutions with appropriate legal authority, financial resources, technical capability, and an interest in the resource are central to lake and shoreland management. In Wisconsin, there are two complementary State-local programs: The Shoreland Protection Program and the Lake Management Program. Their legislative development, their regulatory powers, and the results of their implementation are discussed.

INTRODUCTION

Lake Problems

Increasing development brings problems to many lakes. The amenities of a natural shoreline are replaced by ribbons of structures. Dwellings may be squeezed onto undersized lots too close to each other and too close to the water. With the removal of shoreland vegetation, native plant communities are destroyed and wildlife habitat disappears. Erosion problems also intensify as vegetation is removed. Road building, grading, and filling during development exposes raw earth and causes additional erosion. Silt muddies the water and impairs aquatic life. In some places, municipal and industrial wastes and agricultural runoff are the major polluters. Septic tank systems which serve most recreation developments can add excess nutrients to the lake, if improperly installed or maintained. Other lakes are free from these sources of pollution but need management to protect their quality.

Solutions to Lake Problems

Proper shoreland development can be a key factor in avoiding lake problems. Maintaining a natural strip of shoreline can provide a buffer between land and water. Buffer strips are natural preservers of water quality as they trap nutrients and retard erosion. Preservation of the shoreland buffer also conserves the unique scenic qualities of the lakeshore. Density standards to avoid overcrowding and proper installation of on-site waste disposal systems can also help preserve lakes.

Where problems already exist, certain management practices can, in some cases, prevent further degradation or repair past damage. There are two general approaches to lake protection and rehabilitation: (1) Limiting fertility; and (2) treating the products of overfertilization. Limiting fertility can consist of measures such as diversion, nutrient inactivation with chemicals, and dilution through lake flushing. Treating the products of overfertilization includes measures such as mechanical aeration to increase oxygen levels and mechanical harvesting of excess weeds (Born and Yanggen, 1972).

Institutions

Institutions with appropriate legal authority, financial resources, technical capability, and an interest in the resource are central to lake and shoreland management. In Wisconsin, there are two complementary, State-local programs: The Shoreland Protection Program and the Lake Management Program. The Shoreland Protection Program establishes minimum State standards for mandatory local zoning and sanitary (septic tank) and subdivision ordinances to protect the shoreline environment. Shoreland ordinances which include lands within 1,000 feet of lakes contain, among other things, provisions governing: Minimum lot size and width; waterline setbacks; removal of vegetation; on-site waste disposal, filling, grading, and dredging; and wetland protection.

The Lake Management Program authorizes the creation of special purpose districts at the local level and provides these lake districts with technical and financial assistance from the State. A district has power to tax, levy special assessments, borrow, and bond to raise money. It may make contracts, hold real estate, and disburse money for lake protection and rehabilitation projects, but does not possess the regulatory power.

THE SHORELAND PROTECTION PROGRAM

The Enabling Statute

In adopting sections 59.971 and 144.26 of the Wisconsin statutes, the legislature (1) created special shoreland protection corridors, i.e., unincorporated lands within 1,000 feet from a lake, pond or flowage, 300 feet from a river or stream or to the landward side of the flood plain if this is a greater distance; (2) established special regulatory objectives to protect water quality and shoreline amenity values; and (3) provided that if a county failed to adopt shoreland regulations meeting minimum State standards within 18 months after passage of the law, a State agency was required to adopt the regulations.

To implement the broad mandate of this law, a number of difficulties had to be overcome:

1. The regulations had to be capable of being applied on a statewide basis involving many miles of lake and stream shores;
2. Few counties had modern zoning, subdivision control, and sanitary regulations, or experience in adopting and administering them;
3. There was a lack of detailed resource data on specific shoreland characteristics such as soil types, slope, vegetative cover, land use development patterns, direction of groundwater flow, and water quality parameters, which may vary widely for individual water bodies;
4. Limited scientific data made it difficult to generalize about the potential pollutional effects of various uses;
5. Constitutional limitations on the use of the regulatory powers, such as those that prohibit the taking of private property without compensation had to be considered;
6. It was necessary to strike a balance between the concern with shoreland problems and the desire for economic revenue from shoreland development to maintain political support for the regulatory program;
7. The regulations had to be designed to be feasibly administered and enforced.

The Department of Natural Resources, which was charged with supervising county compliance, was assisted by the University of Wisconsin and State and Federal agencies in preparing a shoreland protection manual and model shoreland protection ordinance. The purpose of these publications was to provide those counties lacking professional assistance with information to help meet the 18-month deadline.

The Shoreland Protection Ordinance

The model shoreland protection ordinance is essentially a natural resource oriented development code (Yanggen and Kusler, 1968). The basic land use controls available to local government, that is, zoning, subdivision regulation, and sanitary codes, are combined in an integrated package. Special provisions not usually found in these regulatory devices are added to meet the special objectives of the shoreland protection law. Many of the regulatory standards are keyed to the physical characteristics of the site. This information is generated at the time of an application for development permission. Certain special uses with potential problems require a case-by-case evaluation by an administrative agency according to standards set forth in the ordinance. The resulting ordinance consists of broad regulations applicable to all shorelands, together with a basic three-district zoning use classification.

Certain controls apply to all shoreland areas regardless of the zoning district in which they are located. These regulations include minimum standards for water supply and waste disposal, tree-cutting controls, setbacks for structures from highways and navigable waters, minimum lot sizes and widths, filling and grading limits, lagooning and dredging controls, and subdivision regulations. These provisions constitute the central core of the recommended regulations.

The manner in which common shoreland uses are developed usually threatens the quality of shoreland areas more than the encroachment of incompatible uses. Typical lakeshore development consists of cottages, residences, and resorts, with occasional taverns, groceries, or other commercial buildings on some lakes. Few recreational areas are threatened by severe nuisance uses like factories or junkyards.

The main problems in shoreland areas are overcrowding, deterioration of water quality, and destruction of shore cover and natural beauty, stemming from: (1) Inadequate lot sizes, side yards, and setbacks from the roads and water; (2) improperly functioning sewage disposal facilities; (3) development practices which lead to extensive erosion; and (4) indiscriminate tree cutting and filling of wetlands. These problems result not so much from the particular use placed on the lot, but from the size of the lot, its suitability for on-site waste disposal, and the manner and placing of development. The basic development code is geared to meet these problems.

Septic systems: The soil of the absorption field is an integral part of a septic tank system and a source of frequent failure. When a system fails, the bacteria-laden effluent backs up into the house or runs out onto the land surface, causing health hazards. This is particularly serious when the effluent reaches a water supply or open water.

Failing septic tank systems and even efficiently operating systems may contribute to a more subtle type of pollution. As septic tank effluent seeps through the soil, filtration of nutrients is incomplete. Depending on the direction of the groundwater flow, these nutrients may enter a lake or river. With these additional nutrients from sewage effluent, weeds and algae may overproduce and cause nuisances. This problem is particularly serious in lakes, which do not have the assimilative capacity of streams.

Since domestic waste disposal is a serious problem in shoreland areas, a sanitary permit for private sewage disposal facilities is required prior to building any structure intended for human occupancy. A permit will not be issued for areas which cannot properly absorb septic tank effluent (that is, steep slopes, high bedrock, high ground water, and impermeable soils) unless these limitations can be overcome. Sites are to be checked for limiting conditions by on-site inspection, including soil borings and percolation tests, and the use of detailed soil surveys where available. Assuming that a site exists with suitable physical properties for soil absorption of liquid wastes, there are additional provisions in the "sanitary code" portion of the ordinance. Detailed standards pertaining to the construction, location, and maintenance of septic tank systems are included.

Tree-cutting: These regulations apply to a strip paralleling the shoreline and extending 35 feet inland from the water. No more than 30 percent of the length of this 35-foot-deep strip may be cleared to the depth of the strip. The cutting of the 30 percent must not create clearcut openings greater than 30 feet in width. In the remaining 70 percent, cutting must leave sufficient cover to control erosion and to screen cars and structures (except boathouses) visible from the water unless a special cutting plan is permitted by the board of adjustment.

Tree-cutting regulations are designed primarily to protect the scenic beauty of timbered shorelines while still allowing a view of water from the lot. There are important secondary benefits. Retaining shoreland vegetation makes land less vulnerable to erosion. Substantial shoreland cover can also reduce the amount of nutrients and other pollutants reaching the water. The shoreland vegetation uses nutrients contained in effluent and fertilizers as food. The vegetation can block other pollutants and debris from entering the water.

Setbacks: In addition to setbacks from highways (typical of conventional zoning) all structures except piers, wharves, and boathouses must be set back from the water. Setbacks help preserve shore cover, natural beauty, and wildlife along the land-water fringe. A 75-foot setback from the water is required of all structures except boathouses. Increased setbacks are recommended for bodies of water that possess outstanding fish and aquatic life, shore cover, natural beauty, or other ecological attributes.

Lot size: A minimum area of 20,000 square feet and minimum width of 100 feet are required for all new shoreland lots not served by public sewers. This is the minimum size considered necessary to achieve other dimensional requirements such as setbacks from the water and roads, separating distances between private sewage disposal facilities and wells or navigable waters, side yards and parking areas; and the shore-cover protection strip along the water.

Filling and grading: These provisions are aimed at reducing erosion from raw soil and controlling filling of wetlands. Land that has surface drainage toward the water and is within 300 feet of a navigable waterway can be filled or graded only by special permit if the exposed area and the slope exceed a minimum figure. The permit must be obtained from the board of adjustment, which can attach a variety of conditions to minimize erosion.

Excavating: Lagooning and dredging provisions are designed to protect wetlands, prevent slumping of sides of excavated areas, and protect fish from oxygen-depleted conditions which may prevail in improperly constructed lagoons. A special permit, contingent upon overcoming these problems, is required for dredging or constructing any waterway or lagoon, or pond within 300 feet of a navigable water.

Subdivision controls: Regulating the division of land into lots for sale is an important part of the shoreland protection ordinance. Percolation tests, soil borings, detailed soil surveys, and other physical data are used to determine that a specified percentage of each lot within the subdivision is free from physical limitations such as impermeable soils, high ground water, near-surface bedrock, excessive slopes, and flooding. Lot size is geared to the degree to which an area is free from a combination of these limiting conditions. The 20,000 square-foot lot area and 100-foot width is the minimum size permitted. Lots with less favorable physical site factors must have a correspondingly larger size. The presence of limiting factors beyond a certain point prohibits subdivision.

Planned unit development: These provisions allow a developer greater flexibility to arrange lots in clusters rather than in long strips along the shore. The minimum lot size for each dwelling can be reduced if an equivalent portion of the subdivision is restricted to permanent open space. Clustering lots on suitable terrain reduces land improvement costs and makes common sewerage and water systems economically feasible. Wetlands, steep slopes, and other difficult-to-develop areas can be preserved as scenic assets. One subdivision in northern Wisconsin is laid out with all residential development in offshore clusters. The entire lake is buffered by undeveloped land extending back 200 feet from the shoreline. This shoreline strip is owned in common by purchasers of residential lots. The residential clusters, in turn, are linked with each other and certain recreational facilities by the shoreland buffer and other commonly-owned greenways. The profit the developer foregoes by not subdividing the high value shoreline property is more than compensated for by the increased value of the more numerous offshore lots. Planned unit development provisions can permit thoughtful design which preserves environmental resources while enhancing property values.

Wetlands: The model approach suggests that all substantial wetlands in regulated shoreland areas be placed in "conservancy districts." Wetlands are defined as areas where ground water is at or near the surface of the ground much of the year. These areas are either delineated on U.S. Geological Survey maps or detailed soil survey maps. Some wetlands along water provide fish spawning grounds, whereas others may be prime wildlife habitat. Wetlands are seldom suitable for building because of septic tank failure, unstable soil conditions, and seasonal flooding. For these reasons, the conservancy district regulations limit building development.

Permitted uses of land in conservancy districts include harvesting of wild crops, forestry, wildlife preserves, hunting, fishing, the display of certain signs, and other uses that do not include residential structures and that have relatively minimal effects on the natural environment. Special exception uses include dams, flowages, removal of topsoil or peat, general farming, cranberry bogs, and other uses that may substantially affect the environment. Filling and drainage are also special exceptions and may be used to overcome the natural development limitations of some of the areas. If the board of adjustment grants a special use permit for filling or drainage, and if the wetland area is made suitable for building development in conformance with the conditions imposed, the county board can then amend the district boundaries to place the area within another zoning district.

Implementation: To take constitutional constraints and informational limitations into account, a number of techniques were used in drafting the ordinance: (1) The objectives of the regulations are set forth in considerable detail in an introductory statement of purpose spelling out the relationship between various objectives and the means used in the ordinance to accomplish them. The rationale of various regulatory sections is elaborated in portions of the ordinance, and wherever possible, the relationship to public health and water pollution control is indicated.

(2) Many of the regulatory standards are keyed to the physical characteristics of the site. This information is generated at the time of an application for development. For example, proposals for all uses involving onsite sewage disposal must be accompanied by detailed information about soil permeability, slopes, depth to bedrock, and height of ground water. This information is based on percolation and soil boring tests conducted by a licensed technician.

(3) Uses which are potential sources of pollution or could have other adverse effects are evaluated on a case-by-case basis. The applicant for a special exception permit must supply detailed information about the proposed use to the county board of adjustment. The board investigates the likely effects of the proposed use and decides whether to refuse, grant, or conditionally grant the special exception permit. Standards for the board's investigation and conditions which may be attached to the permit to minimize detrimental effects are set out in ordinance. If needed, technical assistance is available from field representatives of the Department of Natural Resources, Division of Health, Soil Conservation Service, and other agencies. This combination of detailed standards and availability of technical assistance lessens the likelihood of arbitrary decisionmaking and the attendant danger that a court will find the regulations invalid.

WISCONSIN LAKE MANAGEMENT PROGRAM

Legislative History

Wisconsin's lake management program originated in the Inland Lake Demonstration Project, a joint venture by the Wisconsin Department of Natural Resources and University of Wisconsin-Extension (Born, 1974). The project was designed to demonstrate the technical feasibility of several lake protection and rehabilitation techniques and to examine the institutional capability for using those lake management tools that showed potential for practical application.

After 6 years of testing physical methods for improving water quality in Wisconsin lakes with various characteristics (seepage, flow-through, reservoir) and a review of lake renewal work in other States and countries (Dunst, et al. 1974), project personnel concluded that applied lake management, though still in its infancy, could be carried on in a general management program. Concurrent examination of various units of government (State, county, town, sanitary district) and private groups (lake associations) indicated that no existing institution had both the interest and the legal structure to provide the necessary authority, financing, and long-term commitment for managing lake resources (Klessig, 1973).

The State had neither the staff nor the financial resources to individually manage 9,800 lakes. Counties and towns were more concerned with providing services, such as roads, that permanent residents (voters) demanded. Sanitary districts had bent toward lake management, but their enabling authority to do so was shaky. Lake associations often exhibited strong interest, but as voluntary groups they had no legal authority to manage a public resource (Klessig and Yanggen, 1973, 1975).

The final task of the Inland Lake Demonstration Project was to draft legislation setting up the necessary institutional structure. This legislation became Chapter 33 of the Wisconsin Statutes.

The Enabling Legislation

Chapter 33 provides the legal framework for creation of special-purpose units of government to manage lakes and for State assistance to these lake districts. The people who own property around a lake must initiate information of the lake district under the legislation, and the lake district must be established by an official resolution of a general purpose unit of local government (county, city, village, or town) (Klessig, 1976).

Although the lake district is legally independent, the law provides that a member of the county board (soil and water conservation district) and a member of the local municipal governing body be represented on the lake district's board of commissioners. This overlap of governing bodies is designed to facilitate communication and cooperation among the general purpose units of local government, which retain all police powers, such as zoning; the soil conservation district, which helps landowners retard erosion; and the new lake district, which focuses on water quality management but must also cope with the impact of land use patterns.

Operating under the directives of the annual meeting and through a board of commissioners, the lake district is the functional agency for comprehensive management of a given lake or chain of lakes. It not only develops and adopts plans for managing the lake, but also directs any protection or treatment work and takes on long-term responsibility for the community resource. As a unit of government, the lake district has the full range of powers to make contracts, hold real estate, disburse money, and levy a property tax. Its specific lake management powers include, but are not limited to: (1) Study of the causes of existing or potential lake problems; (2) prevention and control of aquatic weeds; (3) prevention and control of algae; (4) prevention and control of swimmer's itch; (5) aeration; (6) nutrient diversion, removal, or inactivation; (7) erosion control (voluntary cooperation and financial assistance for landowners); (8) dredging; (9) treatment of bottom sediments; and (10) construction and operation of water-level control structures.

The second part of Chapter 33 provides for technical and financial aid to lake districts. This aid is based on the premise that lakes are used by the general public, and the general public should bear a portion of the costs of lake management. (A lake district can only be formed around lakes with public access.) The Wisconsin Department of Natural Resources is the lead agency in providing this aid through its Office of Inland Lake Renewal (Klessig, et al. 1978).

In addition to its new program of assistance to offical lake districts, the DNR retains its statutory responsibility as the trustee of public water. It must approve lake management plans submitted by districts and issue permits for most in-lake activities carried out by districts.

Information for lake property owners and public officials is provided by lake resource management specialists and county-based resource agents of the University of Wisconsin-Extension.

Organizing a Lake District

The process of lake district development usually begins when community leaders attend a regional conference where lake management is discussed by State lake management professionals. If the community leaders feel the program is applicable in their case, they initiate the district formation process. The County Extension Office plans an educational meeting in the community. Often accompanied by a DNR resource manager, a University of Wisconsin-Extension specialist makes a presentation and provides attendees with educational materials and guidelines on using Chapter 33 (Klessig, 1977).

At the conclusion of the meeting, an ad hoc organizing committee of property owners, with the assistance of Extension, Soil Conservation Service and DNR officials, goes about the arduous task of defining a proposed lake district boundary. Depending on the character of the lake basin and political realities, the district may include a narrow strip of cottage lands around a lake or encompass an entire watershed.

The committee then collects signatures of landowners within that boundary. Once the petition has been signed by a majority of landowners, the town board or county board holds a hearing and uses the following criteria from Chapter 33 to decide whether or not to create the district:

1. Has the petition been signed by at least 51 percent of the landowners or owners of at least 51 percent of the land?
2. Is the district necessary?
3. Will the public health, comfort, convenience, necessity, or public welfare be promoted?
4. Will the included property benefit?
5. Will the district cause or contribute to long-range environmental pollution?

Operating a Lake District

Five commissioners govern a lake district. They include three residents or property owners within the district elected at the annual meeting of the lake district each summer; one member of the town board, village board, or city council with the highest assessed valuation (equalized) in the district and appointed by that governing body; and one supervisor of the county soil and water conservation districts (who in Wisconsin is also typically a county board member), appointed by the county board.

The commission applies for technical assistance from the Office of Inland Lake Renewal by compiling all existing information on the lake and its watershed. A feasibility study design is prepared by the centralized, interdisciplinary staff of the Office of Inland Lake Renewal. If the district decides to proceed, it contracts with a private consulting firm to collect additional data as prescribed in the design. A State grant pays for 60 percent of the study cost. The remaining 40 percent is paid by the lake district through its taxing powers and/or by local volunteer efforts that reduce the cash cost of the feasibility study.

The results of the study are returned to the interdisciplinary team in the Office of Inland Lake Renewal for analysis and formulation of alternative methods of protection or rehabilitation. The lake district selects and modifies the alternatives to conform with local values and financial resources. The lake district plan is then submitted to the regional planning commission and the soil and water conservation district for comment and finally to DNR for approval.

Following a public hearing held by DNR in the local area, DNR may approve the plan and provide up to 80 percent funding. Depending on the character and scope of the project, the Office of Inland Lake Renewal may prepare a grant application for the district and submit the proposed project to the Environmental Protection Agency. If Federal funding is also approved, EPA funds 50 percent of the project, DNR 30 percent, and the lake district 20 percent.

The lake district decides at an annual meeting whether to proceed. If a majority of the resident and non-resident property owners present favor implementation of the plan, the district commissioners sign the necessary contracts, and implement a management plan. Simultaneously, Extension provides lake district commissioners with newsletters, handbooks, personal consultation, and workshops on operating the new unit of government (Klessig, 1979).

SUMMARY AND CONCLUSIONS

The Shoreland Protection Program and the Lake Management Program are complementary in philosophy and objectives. Improved water quality is the goal of both programs.

They both involve a strong State role—the shoreland program requires that counties adopt regulations meeting minimum State standards if the county wishes to avoid State level regulations. The lake program involves State participation through financial and technical assistance to lake districts and by requiring State approval of lake management plans.

Both programs also involve a strong local governmental role. The county is the main actor in the shoreland program. All Wisconsin counties have adopted shoreland regulations without direct State intervention. These regulations, some of which exceed State minimum standards, are administered on the county level. The lake program has resulted in the formation of over 120 lake protection districts. These local special purpose districts are undertaking a variety of lake protection and rehabilitation activities.

These State-local cooperative programs reinforce each other. While shoreland regulations are largely a protective measure, the lake program includes both rehabilitation and protection. Rehabilitative activities undertaken include storm sewer diversion, chemical inactivation of nutrients, aeration, dam construction, and dredging. Protective activities include serving as a forum to encourage proper administration of shoreland regulations or upgrading of shoreland standards, monitoring of septic systems through attainment of sanitary powers, cost-sharing for manure storage, and purchase of grass waterways and wetlands.

The Shoreland Protection Program provides a countywide framework for regulating land use at the fragile land and water interface. The Lake Management Program provides the citizens who live near that interface with a mechanism to manage the specific resource that attracted them.

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SAMPLING STRATEGIES FOR ESTIMATING CHLOROPHYLL STANDING CROPS IN STRATIFIED LAKES

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ABSTRACT

The spatial distributions of chlorophyll *a* in Lakes Mendota and Delavan, Wis. were studied during the 1971-72 stratified seasons. Both lakes are calcareous and normally support large epilimnetic chlorophyll standing crops (CSC) composed of planktonic blue-green algae. Profile sampling the central lake station provides a nearly unbiased estimate (perhaps negative by 2 to 4 percent) of lake-average CSC with a long-term coefficient of variation (c.v.)~16 percent. Lake stations at the margins are individually strongly biased for lake-average CSC and have large c.v.'s. The bias terms result from the prevailing southwest afternoon breezes, and intermittent north/northwest front passages affecting Wisconsin. The large c.v.'s reflect the important wind shifts between sampling dates. For Mendota the mean square deviation (squared percent) in CSC increases approximately as $100 d_s^{4\ 3}$ where d_s is the station separation distance (km). Based on this relationship, the c.v. (percent) for lake-average CSC is given approximately by: (1) $7\ a^{2\ 3}n^{-1\ 2}$ for a "simple random sample;" (2) $7\ a^{2\ 3}n^{-5\ 6}$ for a "stratified random sample;" where a is the average length (km) of the lake's major and minor axes, and n is the number of chlorophyll profiles sampled. For Lake Mendota, with n = 9 and a stratified design, the estimated c.v. for lake-average CSC is~4 percent.

Stauffer (1980b) examined wind stress effects on the position-dependent chlorophyll concentration profile and time scales for lateral redistribution of epilimnetic chlorophyll standing crops in eutrophic Lakes Delavan and Mendota in southeast Wisconsin. In this sequel I consider expected mean square sampling errors for several total lake (epilimnion) chlorophyll estimators and the error in estimating changes in lake chlorophyll standing crop over time.

EXPERIMENTAL

The chlorophyll sampling and analytical procedures follow Stauffer et al. (1979). Let C(x,y,z,t) be a point estimate of chlorophyll *a* concentration (mg m^{-3}), CSC(x,y,t) the position-dependent chlorophyll standing crop (mg m^{-2}), and $<CSC>_t$, the lake-average CSC at time t.

RESULTS

Coefficient of Variation for CSC: Effect of station separation distance

I now consider the effect of station separation distance, $d_s = \{(x_2 - x_1)^2 + (y_2 - y_1)^2\}^{1\ 2}$ on the root mean square (RMS) sampling variation between CSC(x_1, y_1, t) and CSC(x,y,t).

I divide this by $<CSC>_t$, expressed as percent, and call this the sampling coefficient of variation or c.v. (CSC). Three classes of d are considered, "local," "regional," and "inter-regional," corresponding to RMS d of 10 5m, 1.25 - 1.40 km, and 3.5 km. The sampling c.v. (CSC) at the "regional" level was estimated for the Mendota Central Basin (CB) during 1971 (17 d.f.) and for all major subregions (Figure 1) in 1972 (26 pooled d.f.).

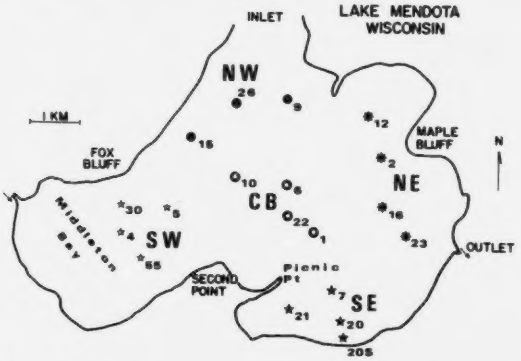

Figure 1. — Sampling stations within identified Lake Mendota sub-regions. CB = Central Basin.

The regional RMS c.v. (CSC) was 12.3 percent, or 9.8 percent if we eliminate the one very large estimate from the southeast lake region on June 30, 1972 associated with a surface bloom (Table 1). The "inter-regional" RMS c.v. (CSC) was 22.6 percent. Individual variates ranged widely (Table 2) but were significantly ($\alpha \leq 0.05$) greater than the regional mean square on 14 of 26 total dates when either Mendota or Delavan were sampled in 1972. Based on 50 pooled d.f. and 14 separate sampling dates the RMS c.v. (CSC) at the "local" level was only 0.6 0.2 percent. The RMS variation in CSC thus increases roughly as $d_s^{2\ 3}$.

Table 1. — Regional coefficients of variation for CSC: Lake Mendota, 1972.

Date	Lake region* and sampling stations		$\langle d_s \rangle$	Mean CSC region	($mg\ m^{-2}$) Lake	c.v.(CSC)°	d.f.'
June 24	NW	9,15	2.10	67	76	6.7	1
	ENE	2,16	1.00	65	76	0.1	1
June 30	SE	20,20S,7	.075	137	102	39.0	1
	CB	10,22	1.25	120	102	11.6	1
	SW	4,5	1.00	77	102	12.9	1
July 8	CB	1,1,1,1,*	1.60	146	149	8.0	3
	ENE	2,16	1.00	172	149	8.9	1
August 6	ENE	2,23	1.40	168	181	1.8	1
August 10	ENE	2,23	1.40	268	239	2.8	1
August 14	ENE	2,23	1.40	199	216	12.4	1
August 22	NW	9,15	2.10	.91	109	0.9	1
	ENE	2,23	1.40	130	109	26.1	1
August 27	NW	9,15	2.10	110	140	17.6	1
	ENE	2,23	1.40	156	140	6.4	1
	SE	20,21	0.60	180	140	4.1	1
September 4	SW	4,30,55	0.80	148	161	4.6	2
	CB	1,6,10	1.40	162	161	6.4	2
	NE	12,2	1.00	159	161	6.2	1
September 21	CB	1,6	1.25	59	57	1.8	1
September 27	ENE	2,23	1.40	51	57	16.8	1
October 7	CB	6,6*	1.60	110	91	10.2	1
October 13	ENE	2,23	1.40	151	133	1.6	1

*CB = Central Basin. $\langle d_s \rangle$ = mean separation distance between stations within region (km); * estimates for 8 July and 7 October CB based on the length of scale, $u^* \Delta t$, for the relevant time period. °Within region. 'd.f. = degrees of freedom.

Table 2. — Coefficient of variation for CSC: Between lake regions: Lakes Mendota and Delavan, 1972.

Lake and date	Number of stations	(mgm^{-2}) lake	$CSC_{max}:CSC_{min}$	c.v.(CSC) (%)	Ratio mean squares'	Significance
Mendota:						
June 10	3	247	1.23	10.7	1.21	—
June 16	3	260	1.28	13.0	1.78	—
June 24	7	76	1.76	22.5	5.36	**
June 30	8	102	3.70	47.0	23.30	**
July 8	7	149	1.57	18.4	7.86	**
July 17 a.m.	3	215	1.33	15.4	2.50	—
July 17 p.m.	4	226	3.67	54.6	31.50	**
July 23	5	194	1.58	24.4	6.29	**
July 24	4	194	1.30	12.3	1.61	—
July 31	4	206	1.42	14.5	2.22	—
August 6	7	181	1.18	6.9	0.50	—
August 10	7	239	1.33	10.2	1.10	—
August 14	7	216	1.49	12.3	1.61	—
August 22	7	109	1.79	26.2	7.25	**
August 27	8	140	1.92	22.9	5.54	**
September 4	9	161	1.43	13.9	2.05	—
September 13	5	138	1.58	17.3	3.17	*
September 21	5	57	1.45	13.4	1.89	—
September 27	7	57	1.67	17.4	3.20	*
October 7	6	91	1.83	20.9	4.62	**
October 13	7	133	1.50	16.0	2.71	—
Delavan:						
June 6	3	105	1.24	10.6	1.18	—
June 20	3	226	2.69	44.3	20.80	**
July 5	3	208	1.42	20.2	4.32	**
July 18	4	268	3.42	54.5	31.40	**
July 30	4	312	1.63	21.6	4.93	**

*Ratio significant at 5% level.
**Ratio significant at 1% level.
'Inter Regional: Grand mean regional (Mendota).

I also examined c.v. (CSC) for all station *pairs* when Lakes Mendota (n 658) and Delavan (n 25) were sampled during 1971 and 1972. Because the air friction velocity, u , is a measure of the surface wind drift current (Hicks, 1972), the alternative metric, $d_s^* = (d_s^2 + u_*^2 \Delta t^2)^{1\,2}$ was also examined because lake sampling could not be made perfectly synoptic (Figure 2). Estimates of u_* were based on algorithms reported by Stauffer (1980a).

Figure 2 reveals a triangular scatter pattern with individual Mendota c.v. (CSC) estimates varying widely at the longer separation distances. Less scatter is evident in the Delavan points. The "no-intercept" (valid because the local c.v. (CSC) estimates $\rightarrow 0$) polynomial regressions for the Mendota and Delavan data are: (note the very small differences between equations 1 and 1a and between 2 and 2a).

Mendota: $n_p = 658$

$$\hat{c.v.}(CSC) = \underset{(0.6)}{8.0\, d_s} - \underset{(0.14)}{0.79\, d_s^2} \qquad R^2 = 0.537 \qquad \text{eq. 1}$$

$$\hat{c.v.}(CSC) = \underset{(0.6)}{7.1\, d_s^*} - \underset{(0.13)}{0.59\, d_s^{*2}} \qquad R^2 = 0.541 \qquad \text{eq. 1a}$$

Delavan: $n_p = 25$

$$\hat{c.v.}(CSC) = \underset{(1.9)}{13.4\, d_s} \qquad R^2 = 0.685 \qquad \text{eq. 2}$$

$$\hat{c.v.}(CSC) = \underset{(1.8)}{12.9\, d_s^*} \qquad R^2 = 0.693 \qquad \text{eq. 2a}$$

(The standard errors of the regression coefficients are in parentheses.) Platt, et al. (1970) also found a rapid increase in c.v. (CSC) for short station separation distances and then little change for sampling quadrants exceeding 2.6 km².

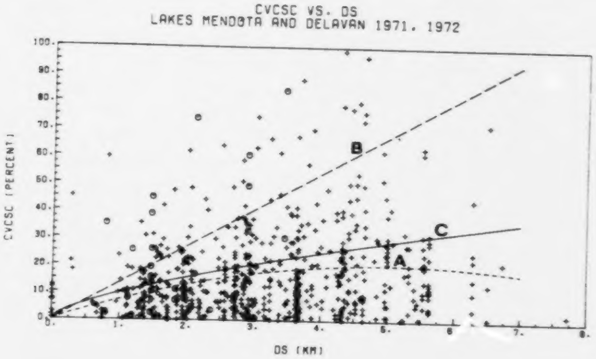

Figure 2. — Variation of c.v. (CSC) vs. d_s for all station pairs: crosses = Mendota, open symbols = Delavan: 1971-1972. A: Best fit "no-intercept" quadratic polynomial (regression Mendota data). B: Delavan quadratic polynomial regression. C: Curve based on d_s relationship inferred from Mendota local vs. regional vs. inter-regional analysis.

Persistent Regional CSC Variations

I now contrast the regional CSC_t values with the lakewide means $<CSC>_t$ for the ensemble of available sampling dates. For the i'th (fixed) lake region and the t'th sampling date, define the statistic z :

$$Z_{it} = 100 \left\{ \frac{CSC_i(t)}{<CSC>_t} - 1 \right\} \qquad \text{eq. 3}$$

Clearly, independent estimates of Z_{it} are generated for each of the dates when the i'th region was sampled as part of a lakewide chlorophyll survey. The statistical moments are:

$$<Z_i> = \frac{1}{N} \sum_{t=1}^{N} Z_{it} \qquad \text{eq. 4}$$

$$\hat{\sigma}_i^2 = \frac{1}{N-1} \left\{ \sum_{t=1}^{N} Z_{it}^2 - N<Z_i>^2 \right\} \qquad i = 1, \ldots, R \qquad \text{eq. 5}$$

If the numerator and denominator of Eq. 3 are kept independent and we approximate c.v. $(1/<CSC>_t) \approx$ c.v. $<CSC>_t$, then, following Goodman (1960), the *long term* sampling standard deviation (in %) for the i'th lake region is approximately:

$$\hat{\hat{c.v.}}_i\,(CSC) \doteq \left\{ \frac{1}{n-1} \sum_{t=1}^{N} \left(\frac{n_t^2 - n_t}{n_t^2} \right) (Z_{it} - <Z_i>)^2 \right\}^{1\,2} \qquad \text{eq. 6}$$

where n is the number of lake station chlorophyll profiles sampled on the t'th date. Finally, the approximate *long term* mean square percentage error (MSE) in estimating lake-average CSC incurred by sampling one station profile in the i'th defined lake region is as follows:

$$\hat{\hat{MSE}}_i\,(CSC) = <Z_i>^2 + \{\hat{\hat{c.v.}}_i\,(CSC)\}^2$$

Z is the "persistent" regional CSC bias of the i'th lake region, expressed in percent.

The five identified subregions of Lake Mendota and the three identified subregions of Lake Delavan have distinct statistical patterns in chlorophyll standing crop (Tables 3 and 4). The Mendota Central Basin stations were very nearly unbiased ($<Z_i> = -2\%$) or CSC (not significantly different than zero even at the $\alpha = 0.2$ level) over the two year span, and also exhibited the smallest regional c.v. (CSC) among the regions tested. Pooling the results for the two Mendota CB stations, and both years, c.v. (CSC) 14.1 percent, based on 62 d.f. The central basin bias for Delavan is also negative (-4.0 percent), but the effect lacks statistical significance. Both the northeast and southeast regions of Mendota are biased positively; the converse is true for the southwest and northwest lake regions. The higher mean CSC levels in the eastern region of Lake Mendota are probably the result of prevailing westerly winds (cf. Stauffer, 1980a, b). The southwest region of Lake Delavan has a negative, and the northeast region a positive, CSC bias (Table 4).

The CSC means of station pairs on opposite ends of the fetch axes (NW—SE, SW—NE) are very nearly unbiased and display a reduced variance. For the 14 dates in 1971 when both the Middleton Bay (SW) and NE stations were visited, the difference between these two stations averaged 43 percent of the lake-average CSC, while the mean of the pair differed from the lakewide mean by an average of only + 2.6 percent. The comparable percentages were 31 and + 1.8 percent for the 11 SE-NW pairs observed in 1971. The calculated mean CSC values for the SE-NW and SW-NE station pairs during the windy afternoon of July 17, 1972 were 228 and 224 mg/m², values which differ from the morning lakewide mean (215 mg/m²) by a maximum 6 percent. Chlorophyll standing crops at the NW and SE stations differ by 270 percent in the afternoon (Figure 4).

Table 3. — Statistical summary of regional CSC departures* from lake-average: Mendota, 1971-1972.

	Central Basin°		Peripheral regions			
Year	D.H.	M.L.	NE	NW	SW	SE
1971 $\langle Z_i \rangle$	-3.3	-5.0	+15.1	-9.8	-11.0	+4.3
$\hat{c.v.}_i$(CSC)	16.8	11.4	26.4	16.7	23.2	23.5
$\sqrt{MSE}$	17.1	12.4	30.4	19.4	25.7	23.9
1972 $\langle Z_i \rangle$	-2.1	+2.8	+2.3	-5.2	-9.7	+13.2
$\hat{c.v.}_i$(CSC)	13.4	14.6	20.3	24.9	12.8	38.0
$\sqrt{MSE}$	13.6	14.9	20.4	25.4	16.1	40.2

*In percent.
°D.H. = Deep hole station. M.L. - Midlake station (Fig. 1).

Table 4. — Statistical summary of regional CSC departures from lake-average: Delavan 1972.

	Percent difference by lake region		
Date	Central (Station 5)	Northeast (Station 7)	Southwest (Station 2)
June 6	-0.4	-10.4	+10.8
June 20	+13.8	+35.8	-49.6
July 5	-13.4	-9.8	+23.2
July 18	-17.3	+53.5	-36.2
July 30	-2.9	+25.8	-22.9
$\langle Z_i \rangle$	-4.0	+19.0	-14.9
$\hat{c.v.}_i$(CSC)	12.2	28.3	31.0
$\sqrt{MSE}$	12.8	34.1	34.4

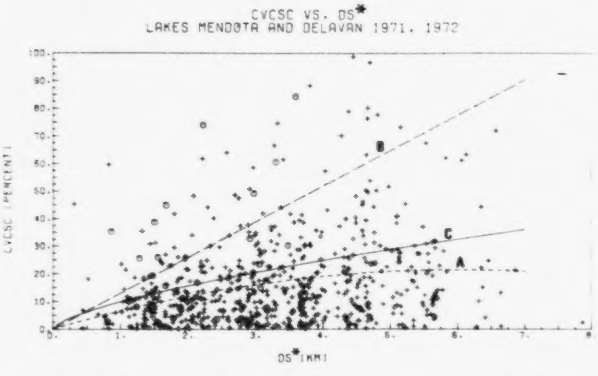

Figure 3. — Lake Mendota chlorophyll spatial distribution: July 17, 1972. p.m.

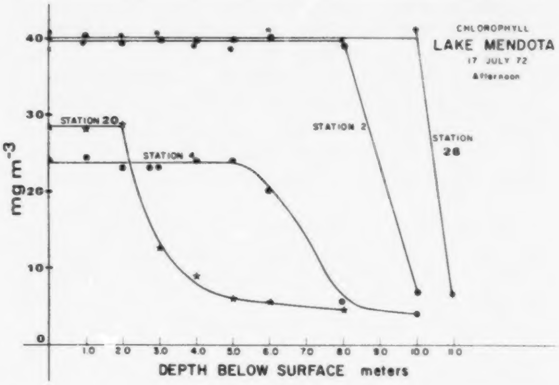

Figure 4. — Chlorophyll profiles for Fish Lake, Dane County, Wis. showing seasonal displacement of chlorophyll maximum to greater depths accompanying stable stratification, and progressive nutrient depletion of the eiplimnion. Fish Lake is a small (A = 0.88 km²), mesotrophic, spring-fed, marl lake with a late-spring-summer epilimnion boundary at 3-5 m, and a summer Secchi transparency averaging 4.2 m. Dissolved oxygen levels are typically highly supersaturated (+5 mg L^{-1}) in the upper metalimnion.

Anisotrophic Character in Inter-regional c.v. (CSC)

A detailed analysis of the events of July 17, 1972 (Stauffer, 1980b) showed that lateral gradients in CSC intensify along persistent wind-fetch axes, i.e., c.v. (CSC) is anisotropically related to d . Thus, station separation distances corresponding to the inter-regional level can feature either extreme differences in CSC (SE vs NW, July 17 p.m.) or modest differences in CSC (SE vs SW, July 17 p.m.). This explains the triangular pattern of the Mendota data points in Figure 2. Less scatter was observed in the Delavan data because of the lake's linearized morphology (cf. Stauffer, 1980b), with sampling stations lying only along the principal wind fetch axis.

Stauffer (1980b) showed that lateral gradients (expressed as percent of $\langle CSC \rangle_l$) increase with $\langle CSC \rangle_l$, probably because large standing crops are buoyance prone, hence susceptible to wind advection. Curve C (Figure 2) shows this relation for hyper-eutrophic Lake Delavan.

Expected error in $\langle CST \rangle_l$

I consider now the expected mean square error in lake chlorophyll sampling. Assume intitially, that detailed information on antecedent weather conditions is either lacking, or that shifts in wind magnitude and direction during the 48 hours prior to sampling lead to indeterminancy in the vector predictor of CSC gradients (Stauffer, 1980b). Hence, we have no conditional expectation of regional differences in CSC. Assume also that the lake is box-shaped with side length, a= 6 km; hence, surface area (36 km²) is only slightly less than that of Mendota (39.1 km²). Recalling that:

$$\text{M.S.(CSC)} \simeq 95\, d_s^{4\ 3} \qquad \text{eq. 8}$$

what are the expected errors in CSC for the following sampling strategies:

1. A simple random sample (srs) of chlorophyll profiles of size n?

2. A stratified random sample (str) of n equal-area squares which partition area a^2? (Readers unfamiliar with sampling theory should consult Cochran, 1963.)

For two points randomly placed within a square of the side length a, the expected (denoted E) mean square distance btween the points can be shown to be exactly $a^2/3$. From this relationship, we can *approximate* E $d_s^{4\ 3}$ as $a^{43}/2.08$, and

$$\text{M.S.}_a\text{(CSC)} \simeq 46\, a^{4\ 3} \qquad \text{eq. 9}$$

where the subscript a denotes that the mean square variation in CSC is for points randomly placed in a square with side length a. The expected error in $\langle CSC \rangle_t$ for a srs of size n profiles is then given simply by

$$\hat{c.v.}_{srs} \langle CSC \rangle_t \approx 6.8\, a^{2\ 3}/n^{1\ 2} \qquad \text{eq. 10}$$

For a = 6 km, the expected error for srs decreases from 22.5 to 11.2 to 7.5 percent as n goes from 1 to 4 to 9.

If we partition the original lake area into component squares and conduct a stratified random sample with one profile per component square, the expected error is

$$\hat{c.v.}_{str} \langle CSC \rangle_t \approx 6.8\, a^{2\ 3}/n^{5\ 6} \qquad \text{eq. 11}$$

In particular, for n = 4,9, the stratified random sample has expected error of 7.1 and 3.6 percent, respectively. As in the case of any commodity which is overdispersed in the sampling space, a stratified random sample gives higher precision than a simple random sample.

A systematic sample can yield improved precision over a stratified sample and is easier to execute. Earlier it was shown that a single chlorophyll profile near the midpoint of the Mendota Central Basin is very nearly unbiased and estimates lake CSC with an RMS error of ~ 16 percent. A systematic sample for larger n can be constructed by sampling the lake's midpoint and station pairs equidistant from the midpoint along the principal and transverse axes. The relative precision of the systematic sample (as compared to a stratified sample) can be expected to increase with increasing magnitude and predictability of the downwind CSC gradient. Cochran (1963) notes that whenever the correlogram for a sampling variate is concave upward, a systematic sample is more precise than a stratified sample; this condition is met here.

The expected error in lake-average CSC was directly estimated for Lake Mendota by comparing the mean CSC estimates for the morning (3 stations) and afternoon (4 stations) of July 17, 1972 and the mean CSC estimates of September 20 and 21, 1971 (3, 5 stations, respectively). Based on 2 d.f., CSC had an uncertainty of 6.0 percent; this includes the effect of time for the two short intervals. The estimated error is gratifyingly close to the prediction of Eq. 11.

If we now assume that the wind has been relatively steady and strong during the 48 hours prior to sampling, CSC(x,y) will increase monotonically with increasing distance downwind along the fetch axis (Stauffer, 1980b). Hence, the conditional expectation of CSC(x,y) is biased positive or negative, depending on whether (x,y) falls in the downwind or upwind region. A single station near the midpoint of the fetch axis will be nearly unbiased for lake-average CSC. A systematic sample involving the midpoint and two symmetrically placed stations out along the fetch axis will provide a strong, nearly unbiased estimate of lake CSC. However, care must be exercised in chlorophyll transect sampling. Significant elapsed time between stations can lead to biases in $\langle CSC \rangle^t$ because of confounding between the diel windpower cycle and the station visitation sequence (Stauffer, 1980a).

Optimal allocation of sampling effort

Expected lateral gradients in CSC influence the optimal density of the sampling stations within separate lake regions. In fact, "Neyman optimal allocation" dictates that the regional station density (stations Km^{-2}) increase approximately as the square of the expected regional CSC (Cochran, 1963).

Until now we have assumed that chlorophyll samples were systematically obtained at integral meter depths below the surface. Under what circumstances should this design be modified? Again by Neyman allocation; sample density (samples m^{-1}) within the profile should increase with the expected gradient, $\partial c/\partial z$. How then do we form our gradient expectations? Clearly, under conditions of elevated turbulence, the vertical spacing of the epilimnion chlorophyll samples can be increased markedly, but only until the epilimnion-metalimnion boundary, h, is approached (Figure 3).

A simple set of decision rules regarding allocation of sampling effort within profiles can be formulated as follows:

1. If oxygen supersaturation occurs with the metalimnion and/or the Secchi transparency depth >0.5 h , then a sharp temperature-dependent chlorophyll maximum can be expected below h (Denman, 1977; Fee, 1976; Figure 4). In such cases, the density of chlorophyll samples along the z axis should be proportional either to the temperature or oxygen gradients.

2. If h 3 X Secchi depth, ignore chlorophyll concentrations below that boundary.

3. Above h, sample proportionally to the temperature gradient.

Inferences concerning changes in lake chlorophyll over time

The variance of the difference of two random variables (Hogg and Craig, 1970)

$$\mathrm{Var}(X - Y) = \mathrm{Var}\,X + \mathrm{Var}\,Y - 2\,\mathrm{Cov}(X,Y) \qquad \text{eq. 12}$$

can be used to estimate the expected error in chlorophyll standing crop changes over any time interval$(t_1 - t_0)$. Under the reasonable assumption that sampling and analytical errors are independent of the sampling dates, the Cov term disappears. If a stratified random sample of size n profiles was obtained on each date, then under the null hypothesis that standing crop was in fact equal on the two dates, equations 11, 12, show that

$$\hat{\sigma}_{\Delta} = \sqrt{2}\{\hat{c.v.}_{str} <CSC>_t\} = 9.6\,a^{2.3}/n^{5.6} \qquad \text{eq. 13}$$

where $\hat{\sigma}\Delta$ is expressed in percent of $<<CSC>>$. For n = 1, a 64 percent change in lake CSC would be required to reject the null hypothesis (time equivalence) at the .05 level. For a systematic sample of n= 1 (Midlake station), approximately a 45 percent change in CSC is required. However, for a stratified sample of size n= 4 on each date, only a 20 percent change in $<CSC>$ is required for rejection of the null hypothesis at the $\alpha = .05$

In some instances, research focuses on documenting chlorophyll response to a natural or cultural lake perturbation. The specific effects of storm-activated nutrient transport or sewage diversion projects are examples. If an extended $<CSC>^t$ record is available, bracketing the "event" date, a time series "intervention analysis" employing a "dummy" variable yields the strongest inferences on the specific intervention effect.

Figure 5 illustrates the change in epilimnion chlorophyll accompanying the powerful July 12-13, 1971 cold front on Lake Mendota (Stauffer and Lee, 1973). $<CSC>_t$ increased from 52 to 111 mg m^{2t}. As n = 5 on both dates, the standard error is calculated to be 7.2 mg m^{-2}, or 12 percent of the estimated change in $<CSC>_t$.

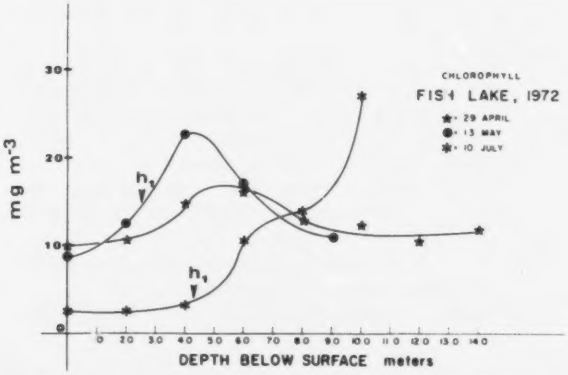

Figure 5. — Lake Mendota vertical chlorophyll distributions: July 8, 14, 1971.

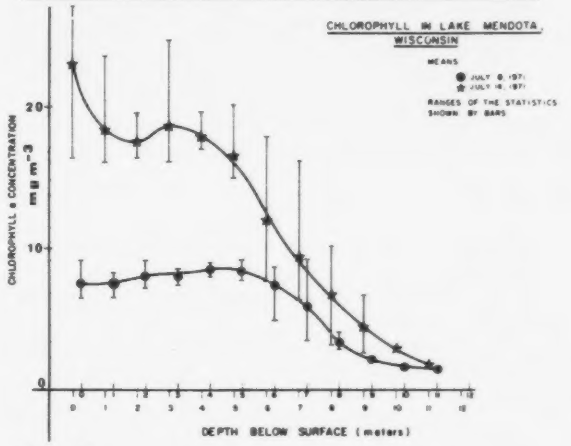

Figure 6

Table 3. — Statistical summary of regional CSC departures* from lake-average: Mendota, 1971-1972.

Year	Central Basin° D.H.	Central Basin° M.L.	Peripheral regions NE	Peripheral regions NW	Peripheral regions SW	Peripheral regions SE
1971 $<Z_i>$	-3.3	-5.0	+15.1	-9.8	-11.0	+4.3
$\hat{c.v.}_{i}$(CSC)	16.8	11.4	26.4	16.7	23.2	23.5
$\sqrt{MSE}$	17.1	12.4	30.4	19.4	25.7	23.9
1972 $<Z_i>$	-2.1	+2.8	+2.3	-5.2	-9.7	+13.2
$\hat{c.v.}_{i}$(CSC)	13.4	14.6	20.3	24.9	12.8	38.0
$\sqrt{MSE}$	13.6	14.9	20.4	25.4	16.1	40.2

*In percent.
°D.H. = Deep hole station. M.L. - Midlake station (Fig. 1).

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THE INFLUENCE OF NUTRIENT ENRICHMENT ON FRESHWATER ZOOPLANKTON

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ABSTRACT

Any relevant change of the environment produces quantitative and/or qualitative effects on the biota. As a consequence, in eutrophicated lakes, variations of the zooplankton biomass and structure may be expected. To solve this problem two approaches have been adopted: (1) Relate the change in zooplankton to the trophic evolution of the environment; and (2) compare the zooplankton characteristics from lakes with different trophic level. Interrelation between phyto- and zooplankton, selective predation by vertebrates and invertebrates and competition between species populations belonging to the same zooplankton association may have a significant influence on the zooplankton structure and biomass. In addition, introduction of toxic substances and manmade change in lake hydrology may also modify the zooplankton association. Consequently, we need more quantitative information on the relationships to estimate the actual influence of eutrophication on zooplankton and to ascertain if zooplankton are directly or indirectly influenced by nutrient enrichment. From comparing several lakes, it is evident that the relationship between trophic level of a water body and structure of its zooplankton seems rather complex. As a consequence, in spite of the great amount of information on this subject it seems practically impossible to use zooplankton to classify water bodies on a trophic scale basis, although useful information on their trophic evolution may be obtained.

INTRODUCTION

Significant modification of the physical and chemical characteristics of the environment causes more or less noticeable changes in the biota. Consequently, modifications in the zooplankton structure and biomass can be expected in a water body during passage from the oligotrophic to the eutrophic stage. For the same reason significant differences should be evident between zooplankton associations living in lakes with different trophic stages. Two approaches are commonly adopted for studying zooplankton modifications following the trophic evolution of a water body. In the first approach the present characteristics of the plankton are compared with those of the past. The second approach consists in comparing the characteristics of zooplankton from lakes with different trophic levels. Another paper (Ravera, 1980a) discusses causes of methodological error and the uncertainty in interpreting the results obtained by both methods. Much information is available on the qualitative and quantitative changes of the zooplankton in lakes during their trophic evolution and the characteristics of the zooplankton from lakes with different trophic levels (e.g., Edmondson, 1969; Brooks, 1969; Patalas, 1972; Bonacina, 1977; Ravera, 1977, 1978).

In spite of this abundant information, several aspects of the problem are still a matter of discussion. In this paper we briefly describe the most important causes affecting variations in the zooplankton biomass and structure in eutrophicated lakes (e.g., changes in phytoplankton density and species composition). The opinions of different authors on the most important points of this problem are compared, and some examples of zooplankton changes in relation to the nutrient enrichment of the lake are reported.

INFLUENCE OF PHYTOPLANKTON

It is generally accepted that nutrient enrichment modifies phytoplankton structure and increases its biomass and production. As a consequence of quantitative and qualitative changes in the phytoplankton, variations can be expected in the structure and biomass of the zooplankton.

This apparent discrepancy between the conclusions of certain authors (e.g., Patalas, 1972), who noted that the increase of zooplankton is related to that of phytoplankton, and those of others (e.g., Nelson and Edmondson), 1955) who observed a great increase of the phyto- but not of the zooplankton can perhaps be justified by the following considerations. A relationship between phytoplankton and zooplankton production can be found only where algal production is a limiting factor. According to Poulet (1978) an increase of zooplankton is limited by the food supply, if this is scarce or if consumption is greater than 50 percent of phyto-production. In several productive lakes, a considerable quantity of the phytoplankton cannot be used by zooplankton because the production of the first is excessive, and zooplankton increase is limited by other factors. The aliquot of phytoplankton not eaten by herbivores passes into the detritivorous chain and accumulates to some extent in the sediments.

Information is scarce on the importance of the organic suspended particles in the diet of zooplankton of lakes with different trophic level; but it is clear that when algal production is low, zooplankton can use organic detritus as a food (e.g., Nauwerk, 1963; Heinle and Flemer, 1975). This change in the diet may mask the relationship between producers and consumers. An increase in predator pressure, reported for several eutrophic lakes, may neutralize the production of zooplankton previously enhanced by the increase of primary production (Brooks, 1969).

In addition, zooplankton variations should be more closely related to variations in the amount of algae suitable as food than to the total phytoplankton biomass. For example, the high correlation between phyto- and zooplankton production found by Hakkari (1978) resulted from the high percentage of algae suitable for zooplankton.

With increasing trophy the percentage of small algae decreases and they are partially replaced by large ones (e.g., Pavoni, 1963; Gliwicz 1967). According to Nilssen (1978) large filter feeders (Daphnids) preferentially use nannoplankton and filtration is more difficult if large algae are present. Large algae are easily ingested by grazers (e.g., Cyclopoids) while the smaller zooplanktons (Rotifers and Crustacean larvae) select nannoplankton from a suspension of algae of different sizes. As a result, large filter-feeders dominate in oligotrophic lakes, whereas small zooplankters and Cyclopoids are abundant in nutrient-enriched lakes. Hrbaceck, et al. (1961) observed that the large filter-feeders (e.g. *Daphnia*) have a more efficient filtration rate than the smaller ones (e.g., Rotifers). Therefore, the large filter-feeders are more favored than other zooplankters in water with low phytoplankton density (oligotrophic lakes).

Interesting results have been obtained by Poulet (1978) from experiments on feeding five marine copepod species. They select particles in relation to abundance and taste but not size. Because of this behavior population growth may be limited by food, but interspecific competition grows stronger. These results seem to contrast with those of other authors. For example, Hutchinson (1951) justifies the presence of more than one Diaptomid in the same lake with the different size of the algae selected by each species according to its body size.

From experiments carried out on three species of *Daphnia*, Korinek (1978) concludes that the duration of the preadult stage decreases with increase in the algal concentration. According to this relationship a balance could be attained between primary production rate and Cladoceran biomass. Consequently, the growth rate should be greater in eutrophic than in oligotrophic lakes under similar thermal conditions. Goulden, et al. (1978) observed that larger Cladocerans have a higher fecundity than the smaller and these, in turn, a shorter growth rate than the former.

On this basis, and considering that fecundity increases with food density, the author concluded that, in the absence of predators, *Daphnia* should be dominant in water bodies rich in phytoplankton, and *Bosmina* in those with a low algal concentration. This interesting hypothesis will have to be tested by further investigation because some authors do not agree on the faster growth rate in *Bosmina*. In *Bosmina, Ceriodaphnia*, and *Diaphanosoma* a slower growth rate has been measured than in *Daphnia* (Novakova, et al. 1978).

While the biomass and the quality of phytoplankton influence zooplankton, they, in turn, influence phytoplankton by their grazing as well as by regenerating nutrients by excretion (e.g., Harris, 1959). In some lakes nutrient regeneration assumes great importance. For example, the quantity of phosphorus excreted by zooplankton in Lake George (East Africa) amounts to 862 tons/year and that of nitrogen to 3,212 tons (Ganf and Blazka; 1974). The ecological importance of P-release by zooplankton and the indirect control exerted on this process by fish predators are discussed by Bartell, et al. (1978).

INFLUENCE OF FISH

It is well known that an abundance of food produces increases in the numbers and growth rate in fish which feed on zooplankters. The effect of fish predation on zooplankton has been the subject of several studies (e.g. Hrbacek, et al. 1961; Brooks, 1968; Hutchinson, 1971; Stenson, 1972).

Because planktivorous fish capture their prey visually, they select in relation to size but not to numbers (e.g. Giussani, 1974). The same fish species may well vary its diet with the season; for example, *Coregonus* sp. from Lake Maggiore prefers *Daphnia* from April to July and *Bythotrephes* from August to November (Giussani, et al. 1977). Although Cyclopoids are not generally considered to be prey for fish (Allan, 1976), in some lakes and seasons copepodits and adults are preyed upon by salmonids (Klementsen, 1968; Jacobsen, 1974).

The influence of predation on the body size of the zooplankters has been clearly demonstrated by Galbraith (1966) in a study on a small lake in Michigan. No fish lived in this lake for 4 years and *Daphnia pulex* was abundant and reached 3 mm length; 4 years after the introduction of fish, the size of *Daphnia* decreased to 1.5 mm. Because these small individuals cannot reproduce, *Daphnia pulex* has been completely replaced by the smaller *Daphnia galeata, D. retrocurva, Bosmina* sp., and small copepods. Another example has been given by de Bernardi and Giussani (1978) for the eutrophic Lake Annone (northern Italy). Mass fish mortality occurred in one of the two basins composing this lake but not in the other. After about 1 month, small zooplankters dominated the one basin with fish; in the basin without fish the most frequent filter-feeder (*Daphnia hyalina*) increased its size from less than 1 mm to more than 1.6 mm in length. In 10 Norwegian lakes Langeland (1978) observed that a high frequency of arctic char (*Salvelinus alpinus*) seems to have a noticable effect on the large zooplankton prey: for example, *Daphnia galeata* and *Holopedium gibberum* decreased their body size significantly.

According to some authors (Hrbacek, 1962; Brooks and Dodson, 1965) intensive predation by fish may completely eliminate larger zooplankton species, while the smaller ones become more frequent. In water bodies without fish, large zooplankters may dominate because their filtration rate is higher than that of smaller zooplankters.

Because fish generally prefer *Daphnia* to Calanoids of the same size, other characteristics of the zooplankton must influence food selection. Zaret, et al. (1976) observed that in a tropical lake, fish preferred smaller and less numerous Cladocerans (e.g. *Bosmina*, *Ceriodaphnia*) to larger and more abundant *Diaptomus gatunensis*, but in the laboratory the same fish ate *Diaptomus* in almost the same amounts as Cladocerans. The authors conclude that in natural conditions the fish feeds only in the surface waters and *Diaptomus* migrates towards this layer during the night, when the fish cannot see potential prey.

In a temperate lake the same authors observed a similar migration behavior in *Daphnia galeata*, probably aimed at reducing predation. Jacobsen (1974) noted a similar strategy in *Megacyclops gigas*. It is clear that in these examples the behavior of the prey protects it more efficiently than its size.

From experiments on *Daphnia pulex* Jacob (1978) concludes that fish predation rate depends upon the combined effect of prey size and light intensity. Some very interesting preliminary results have been obtained by Northcote, et al. (1978) who introduced fish into two small Canadian lakes (Eurice and Katherine) in which there were previously no fish but abundant *Chaoborus*. The substitution of *Chaoborus* by fish seems to have more evident effects on the body size of the prey than on zooplankton density and composition. For example, *Bosmina*, which is not eaten by fish, increased its mean body size, because in the absence of *Chaoborus* larger individuals became more frequent. Among the 10 zooplankters the fish preferred the larger specimens of *Diaptomus kenai* and *Daphnia rosea*, leading to a decrease in the mean body size of these species.

In spite of this, the authors do not believe that fish predation is the most important cause of a reduction of prey body size. Indeed, in late summer, when fish predation was negligible, the body size of the two species remained small and in other seasons, when predation was very active, the number of fish was too low for elimination of all the large specimens. In addition, the body size of the crustaceans preyed upon was very similar in both lakes, in spite of the greater abundance of fish in Lake Eurice. In conclusion, the increase in the frequency of juvenile stages of *D. kenai* and *D. rosea*, an effect of the elimination of *Chaoborus*, is the most probable cause of the body size reduction. Heisey, et al. (1977) observed that fish predation on *Daphnia galeata* and *Daphnia magna* should be more active in the euphotic zone and, consequently, the larger and/or more pigmented zooplankters are more vulnerable than the smaller and transparent ones. In the layers in which light is attenuated and oxygen concentration is moderate, zooplankters, which synthesize hemoglobin at lower concentrations of oxygen (and are consequently transparent), will be less subject to predation than others. These considerations indicate that in an eutrophicated environment the partial or complete substitution of one species by another may be due to several causes and their combinations.

Since fish may have a considerable influence on zooplankton, any variation in the behavior or frequency of these predators is reflected in the zooplankton. In eutrophic lakes, the population density of a coarse fish increases at the expense of salmonids and coregonids (Larkin and Northcote, 1969). For example, in Lake Constance, during the last 50 years the total amount of fish has tripled and in Baldeggersee (Switzerland) around 1940, the white fish were replaced by perch, carp, and pike. New species of fish may modify the predation on zooplankton. In addition, some planktivorous fish, because of an increase of plankton biomass and/or other causes (i.e. reduction of macrophytes) prey upon pelagic zooplankton (Quartier, 1965; Pignalberi, 1967). Any cause producing high mortality in fish (i.e., industrial effluents, water acidification) reduces the predation pressure on zooplankton. A decreased predation may also be caused by fish diseases that are more frequent in eutrophic waters than in oligotrophic. For example, in some eutrophic lakes of Northern Italy mass mortality of bleak (*Alburnus alborella*) due to branchiomycosis, has been frequently observed. This infection seems to be favored by high concentrations of un-ionized ammonia often occurring in eutrophic waters (Giussan et al. 1976).

PREDATION EFFECT BY INVERTEBRATES

The relationships between invertebrates, predator and fish, show a double aspect. In other words, an invertebrate predator may be used as food by fish or they may compete for food. Kajak, et al. (1970) calculated that in two Polish lakes *Chaoborus* larvae daily remove 7 and 13 percent of the total zooplankton. In Leechmere, 94 percent of the *Daphnia* mortality was caused by *Chaoborus* predation (Dodson, 1962). Gliwicz, et al. (1978) estimated that in some water bodies predation pressure by fish is far less important than that of *Chaoborus*, Crustaceans, and Rotifers.

Size selection by fish is the opposite of that by invertebrate predators, because the first prefer larger zooplankters and the latter the smaller (Landry, 1978). For example, Brandl, et al. (1978) observed that *Cyclops vicinus* and *Mesocyclops edax* prey selectively upon smaller zooplankters, such as Rotifers.

Consequently, where predation by fish is severe, small zooplankton dominate, whereas larger forms could be abundant in lakes dominated by invertebrate predators.

If this is true, the abundance of large or small zooplankters is controlled more by the nature of the predation than by the trophic evolution of the water body. For example, O'Brien (1975) observed that in those lakes of the Noatak drainage basin (Alaska) in which there were no fish but a Calanoid predator (*Heterocope septentrionalis*), there were no small zooplankton. Fish prey visually, whereas zooplankton predators use mechanoreceptors and chemoreceptors. Consequently, the prey may escape from plantivorous fish by migrating into deep layers (Zaret, et al. 1976), but by means of alternative behavioral responses (i.e., "dead-man" response) and morphological structure (i.e., spines) zooplankton prey protect themselves from zooplankton predators (Friedman, et al. 1975). Stenson (1976) observed that the ratio of *Bosmina coregoni* to *Bosmina longirostris*, calculated for eight small lakes, is controlled by fish as well as by other predators (e.g. *Chaoborus, Cyclops, Bythotrephes*), which, in their turn, are preyed upon by fish. A study by Dodson (1975) of the predation behavior of 12 zooplankton species revealed that each predator was preyed upon by some other species. These examples and others clearly demonstrate that prey-predator relationships are generally complex.

Zaret (1978) divided predators into three classes: (1) gape-limited predators (GLP), planktivorous fish that select visible prey; (2) size-dependent predators (SDP), invertebrates that prey upon small zooplankters. The third class arises from the coexistence of GLP and SDP.

The zooplankters may escape from predators of the third class by means of their small size, and morphological structures which make capture more difficult. In water bodies, in which predation is the most important factor controlling the zooplankton structure, the dominant forms should be *Ceriodaphnia, Bosmina, Diaphanosoma*, and small *Diaptomus* if GLP are abundant. In the presence of abundant SDP *Daphnia pulex* and large *Diaptomus* could attain high density values. With predation by both GLP and SDP, *Daphnia galeata, Holopedium gibberum* and *Ceriodaphnia lacustris* could be the most frequent species.

CHANGES IN ZOOPLANKTON STRUCTURE AND DENSITY

Lake Lugano. This deep lake, lying between Switzerland and Italy, is now heavily eutrophicated (Ravera, 1980b), but the first effects of nutrient enrichment were observed more than 30 years ago (Baldi, et al. 1949; Jaag, et al. 1970). Relevant variations in zooplankton structure have occurred during the last decades (Ravera, 1978, 1980a); *Eudiaptomus padanus, Mixodiaptomus laciniatus*, and the genus *Sida* have been eliminated and population density of *Daphnia obtusa* has increased. Because Cladocera and Rotifers have no Diaptomid competitor, they have probably attained higher population density than in the past, but due to the different sampling method the reliability of a comparison between our data and those of the preceeding authors is rather small. In addition, the abundance of *Daphnia hyalina* could indicate a moderate fish predation. Our finding agrees with McNaught (1975), who explains the succession of the Cladocerans to the Calanoids with the better pre-adaptation of the former to the eutrophic environment and particularly to the algal size and frequency.

Lake Maggiore (Northern Italy). In 1950 this deep lake was oligotrophic. The first *Oscillatoria rubescens* blooms were observed in 1967 (Ravera, et al. 1968) after a heavy bloom of *Tabellaria fenestrata* had occurred some years before. From 1909 to 1973 the considerable increase of *Daphnia hyalina* and *Chydorus sphaericus* and the decrease of *Mesocyclops leuckarti* seem to show the increasing eutrophication of this lake. During the last 35 years Diaptomids decreased from 40 to 37 percent and Cyclopids from 23 to 8 percent; consequently, the Cyclopoids/Diaptomids ratio diminished (Bonacina, 1977). This is not in agreement with Patalas (1972), who observed a reduction of the population size of Calanoids and an increase of Cyclopoids. The change in zooplankton structure was more rapid in Lake Lugano than in Lake Maggiore probably because the eutrophication rate is slower for Lake Maggiore as compared with that of Lake Lugano.

Lake Mergozzo (Northern Italy). Until 1967 this deep lake was a water body moderately enriched by nutrients. In 1969-1970 blooms of *Oscillatoria rubescens* were observed (Zutshi, 1976) and some modifications of the benthos structure testified to the progressive eutrophication of this lake. This seemd to be caused by an increase of the nutrient loading and the temporary increase of the hydrological renewal time. During 1975 the chemical characteristics of the pelagic waters and those of the phytoplankton demonstrated that this lake has returned to a mesotrophic stage (Saraceni, et al. 1978). From a comparison of the data on zooplankton collected from 1949 to 1975, a significant increase in the population density of Copepods, Cladocerans, and Rotifers is evident. In addition, *Sida cristallina, Mixodiaptomus laciniatus*, and some Rotifers disappeared, whereas *Daphnia hyalina* increased and *Mesocylops hyalinus* has been almost completely replaced by *M. leuckarti* (De Bernardi, et al. 1978). The increase of total zooplankton density from 1949 to 1970 may be caused by nutrient enrichment (Ferrari, et al. 1976).

The increase of zooplankton density from 1970 to 1975 (in spite of an improvement in the conditions of the lake) is not clear and may be the effect of the delayed response of the zooplankton to the decrease of nutrient concentration in the pelagic waters. A strict relationship between trophic level and zooplankton density and biomass has been found by Godeanu (1978) in several lakes of Northern Germany.

Lakes of Brianza (Northern Italy). These shallow lakes (Annone, Oggiono, Alserio, Pusiano, Segrino, and Montorfano) have a high trophic level, except for the latter which may be considered oligo-mesotrophic. From the data reported by Bonomi, et al. (1967) and Gerletti and Marchetti (1977) the following conclusions can be drawn: (1) From 1954 to 1972 the population density of Copepods increased in Lake Segrino and decreased in Lake Motorfano; (2) in Lakes Alserio, Pusiano, and Segrino the total zooplankton density was higher in 1972 than in 1957 but lower in Lakes Annone, Oggiono, and Montorfano; and (3) the population density was roughly in agreement with the nutrient enrichment.

It is rather difficult to explain the changes in population density from 1954 to 1972 as being the effect of nutrient load. Indeed, the changes in population density did not follow a trend, except for Lake Segrino and Lake Montorfano. On the other hand, there is no evidence that in Lake Montorfano this decrease of nutrient load occurred to justify the decrease of zooplankton. For the same reason it is rather difficult to explain the considerable reduction in zooplankton density, particularly of Copepods, during 1967. It is probable that the changes in zooplankton density from 1954 to 1972 are long-term fluctuations controlled by normal meteorological conditions. This hypothesis agrees with the conclusion drawn by

Edmondson (1972) for Lake Washington. In this lake no significant change was evident in the zooplankton structure, in spite of the considerable nutrient load received over a long period of time. As far as the increase of zooplankton biomass is concerned, Beeton (1965) and Patalas and Salki (1973) observed that this increase occurs in some eutrophicated lakes, but it cannot be generalized. Relevant quantitative and qualitative modifications in the zooplankton have been observed during the trophic evolution of Lake Nemi (Central Italy) by Stella, et al. (1978). These changes are partly due to the massive fungal epidemics affecting zooplankton as well as phytoplankton and fishes.

A clear example of the influence of the nutrient loading from domestic and agricultural effluents on the zooplankton is given by Lake Valencia (Venezuela) (De Infante, 1978). From 1968 to 1976 the population density of zooplankton almost tripled, probably because of an increase of phytoplankton of a factor of 10^3. This increase varied with the taxa; a conspicuous increase of Rotifers and significant decrease of Cladocerans were observed. Predation by fish and *Chaoborus* most likely reduced the Cladoceran density. In addition, *Notodiaptomus venezolanus* has been replaced by *Thermocyclops hyalinus*.

INDICATORS OF TROPHIC CONDITIONS

Some zooplankton species have been identified as indicative of very productive water bodies (e.g. *Bosmina longirostris* (Brooks, 1969); *Daphnia cucullata, Acanthocyclops bicuspidatus* (Dussart, 1969); *Brachionus angularis, Brachionus calyciflorus* (De Beauchamp, 1965). Pejler (1957) proposed a list of Rotifer indicators of different trophic stages for Swedish lakes.

According to McNaught (1975) oligotrophic waters are more suitable than eutrophic for *Diaptomus* because of their high filtration and ingestion rate at low density of nannoplankton. There are some exceptions to this statement; for example, *Eudiaptomus padanus* may attain a high population density in very eutrophic lakes (e.g. Lake Varese and Comabbio) and *Eudiaptomus gracilis* is distributed in both eutrophic and oligotrophic lakes. *Mixodiaptomus laciniatus* seems to be more sensitive to eutrophic conditions. In fact, this species has completely disappeared from Lake Lugano, in which it was very abundant when this lake was oligotrophic (1947), and its percentage was decreased considerably in Lake Maggiore because of progressive nutrient enrichment (Tonolli, 1962). The same species disappeared from Lake Mergozzo (Northern Italy) from 1970 to 1975 for the same reasons (de Bernardi, et al. 1976). In oligotrophic waters, *Bosmina* generally have low population density because of comparatively low filtration-rate, but its small size, high birth rate, and capacity to feed on algae of different size, favor its diffusion in eutrophic waters (McNaught, 1975). The same author observed that *Daphnia* occurs both in oligotrophic and eutrophic lakes, because of its high birth rate and filtering capacity and because it can ingest algae of different sizes. On the other hand, its large size facilitates fish predation.

In many cases it is rather difficult to classify the degree of trophy of a lake only on the basis of a list of the species living in it. For example, some Rotifers considered indicators of eutrophicated water (*Keratella quadrata*) have also been collected from oligotrophic lakes. This discrepancy may be due to the different ecogenotypes composing these species (Hutchinson, 1967). The well-known examples reported by Minder (1938) for Lake Zurich and by Deevey (1942) for Linsley Pond (Connecticut), demonstrate that progressive enrichment of the water bodies causes replacement of the larger *Bosmina coregoni* by the smaller *B. longirostris*. According to Hutchinson (1967) the presence of *Bosmina longirostris* in productive lakes and of *B. coregoni* and *B. longispina* in the less productive ones is not an absolute rule.

This has been demonstrated by Findenegg (1943), who collected *Bosmina longirostris* from the epilimnion and *B. coregoni* from the hypolimnion of the same lake. In addition, a rich population of *B. coregoni* (highest value attained is 38,000 individuals/liter) has been studied by Dumont (1967) in the very eutrophic Lake Donk (Belgium). Because the substitution of *B. coregoni* with *B. longirostris* does not seem to be a general rule, Mikulski (1978) proposed to apply the ratio *Chydorus/Bosmina* for estimating the trophic stage of the lake. The value of this index of eutrophication increases with the trophic evolution of the water body. For example, its value for mesotrophic lakes is about 0.5 whereas it ranges between 1 to 7 for eutrophic.

For some species we need sophisticated taxonomical identification. For example, *Cyclops scutifer scutifer* lives principally in oligotrophic waters, while *C. scutifer wigrensis* is a typical form of meso- and eutrophic lakes (Hutchinson, 1967). To evaluate the relationship between the trophic evolution of a water body and the zooplankters inhabiting it, all available information on the ecology and the recent history of the lake should be taken into account.

DISCUSSION AND CONCLUSIONS

Algal growth nutrients cannot directly influence zooplankton, because they are not able to use mineral nutrients as phytoplankton do. On the other hand, nutrient concentration is normally too low to be toxic for most zooplankton species. Nutrient enrichment indirectly influences zooplankton in different ways: for example, by increasing the primary production, varying the biomass and composition of phytoplankton, or depleting hypolimnetic oxygen. This represents the fundamental difficulty in establishing a clear relationship between trophic evolution and zooplankton density and structure.

For the same reason phytoplankton seems to be a more useful indicator of the trophic conditions than zooplankton. In a stable water body zooplankton maintains constant structure, biomass, and production, obviously with seasonal variations and annual fluctuations. As a result of natural causes and anthropogenic influences, these ideal conditions are not the rule, but the exception. Qualitative and quantitative modifications of the zooplankton are often evident. The abundance and structure of the zooplankton are controlled by food, predators, and parasites, in addition to the physical and chemical factors. Therefore, to survive, zooplankton species must develop a complex strategy, which consists essentially of adapting itself to the available food, competing successfully with other species, and protecting itself against predators. When the physical environment is modified by eutrophication, the zooplankton strategy must generally be changed because of variations in the predation pressure and food produced by the trophic evolution. Any species which cannot adapt itself to the changed conditions — or whose adaptation turns out to be inefficient — is eliminated or reduced in number and biomass.

When a species is eliminated because its food supply has become too scarce, its niche disappears; but if the species is eliminated by predation, its niche may be occupied by other species better protected against predators. In addition, the trophic evolution favoring some phytoplankton species (recently immigrated or previously rare), may create new niches available for new zooplankton species.

The relative importance of predation, feeding, and competition varies in different communities. Consequently, all the hypotheses proposed to explain the modification of the zooplankton abundance and composition are probably reliable, but their importance varies with the lake under consideration. Indeed, in spite of the various studies on this subject very few results can be generalized, with the exception of an increase in the zooplankton frequency and biomass observed in eutrophicated lakes and the influence of the predator on the body size of the prey.

There is general agreement on the latter relationship, but more hypotheses have been proposed for identifying the mechanisms responsible for it. Several authors have observed that fish select large prey and may eventually eliminate the prey. It would be interesting to know the fate of the predator fish following the extinction of the large prey. We may imagine that the extinction of the fish follows that of the prey, or the fish may switch its predation from the large forms to the small. If the first hypothesis is true, fish population density should decrease simultaneously with the decrease of the prey, but before the latter are gone. Unfortunately, quantitative information on this subject is almost absent. If the second hypothesis is reliable it seems rather unlikely that the fish eat large prey exclusively. It seems more likely that the fish, with the decrease of its preferred prey, introduces into its diet an ever-growing percentage of small prey. If this is true the reduced predation pressure could permit an increase in the density of large prey in agreement with the Volterra-D'Ancona model. This increase would have to be more rapid and consequently more probable, in those large parthenogenetic prey species having a high intrinsic rate of natural increase such as *Daphnia*, than in the small Cladocerans. When there is no previous information, the absence of large zooplankters in water bodies rich in planktivorous fish, does not necessarily demonstrate that fish have eliminated the large zooplankters, but that, under certain circumstances the fish diet may consist only of small forms.

The problems concerning predation and competition require more information, but our knowledge of the interactions between these processes is even scarcer. One of the few examples is given by Bossone, et al. (1954) on the competition between two Diaptomids and the predation on them by *Heterocope saliens* in a small alpine lake (Northern Italy). In addition, studies in natural and semi-natural conditions, e.g. "micro-ecosystems",nfluence of zooplankton on phytoplankton exerted by grazing and nutrient mineralization are fundamental for understanding the interrelationships between nutrients and zooplankton changes.

In several cases it is rather difficult to discriminate between the effects on the zooplankton structure and biomass caused by eutrophication and those produced by other causes interfering with the trophic processes (i.e., industrial and mining pollution, introduction of new species, overfisning, yearly meteorological changes). As a result, one may suppose that eutrophication is the only cause of zooplankton modifications if these show a well-defined trend over a series of years, and if there are no other apparent influences acting upon the zooplankton.

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USING TROPHIC STATE INDICES TO EXAMINE THE DYNAMICS OF EUTROPHICATION

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ABSTRACT

To use trophic state indices solely for lake classification overlooks their greater potential as diagnostic aids in examining relationships between various factors in lakes. Carlson's index (1977) uses simple models and regression relationships to provide correlated index values for chlorophyll, transparency, and total phosphorus. The values from these three commonly used variables are transformed into a common scale. Frequently, deviations of these variables provide more information about the lake than does their coincidence. The index is used to identify possible nitrogen-limitation, non-algal turbidity, and the impact of zooplankton grazing.

INTRODUCTION

The concept of trophic state has proved to be a durable and useful concept in limnology, although it is plagued with lack of definition and of measurement. Various indices have been proposed to aid in evaluating trophic state. These indices, by one method or another, attach a label or number to the lake, thus classifying it. Although trophic classification is certainly an important aspect of limnological investigations, the pinning of a trophic label on a lake can actually obscure important differences among lakes within that classification.

A unique classification index devised by Carlson (1977) derives index values which calculate the index individually for three variables: Chlorophyll, Secchi disk transparency, and total phosphorus. Under "normal" circumstances all three index values should be similar. Although the values derived can certainly be used to classify lakes, a far more important use of the index has been to examine deviations of index values from those predicted by the other values. These deviations can reveal basic differences in the ecological functioning of certain aquatic systems. This paper illustrates this use of the index.

DATA FOR LAKE SURVEYS

Often agencies are confronted with data reduction and interpretation of large amounts of data from surveys of many lakes. Typically, these surveys have limited numbers of samples, taken at different times of the year. Interpretation of such data sets is difficult because of the lack of "normal" references with which to compare the data. The trophic state index can provide such a reference line, as it is based on the assumption that the index values should be the same in the same samples. Obviously "normality" in this case depends on the original data set used in formulating the index's regression equations, but extensive use of the index on other lakes suggests coincidence of the three variables in common, and that reasons do exist for consistent deviations.

To illustrate the utility of the index in checking large data sets, I used data collected in the National Eutrophication Survey (U.S.EPA, 1978a, b, c). The sets consisted of data from 105 natural lakes and 386 reservoirs located in the southern, central, and western United States. Most of the lakes had been sampled three times during 1 year and the data I used was the median or average value obtained from these three samples. Trophic state index values were calculated from the data for mean Secchi disk transparency, median total phosphorus, and mean chlorophyll *a*. The calculated index values were compared two at a time to a 1:1 index correspondence line based on the assumption of perfect correlation of the indices.

The first comparisons indicated that impoundments showed different relationships among the indices than did natural lakes. For this reason, natural lakes were compared separately from impoundments.

The correspondence between the chlorophyll index and the transparency index in natural lakes is quite good (Figure 1), although there are some outlying lakes. The index comparison quickly identified deviant lakes, which could then be examined more intensively. When the total phosphorus and transparency indices are compared (Figure 2), however, there is a systematic deviation of a large number of lakes, especially at high total phosphorus values. The asterisks indicate that all of these lakes have a total nitrogen to total phosphorus ratio of less than 15:1 by weight. This same trend is seen in the total phosphorus-chlorophyll comparison (Figure 3). Lakes with low nitrogen to phosphorus ratios seem to show significant deviations of the algal indicators from the total phosphorus.

The effect of TN/TP ratios is best seen in Figure 4, where the difference between the chlorophyll and phosphorus index is plotted against the nitrogen-phosphorus ratios. Lakes with a TN/TP ratio of less than 26:1 by weight have significant deviations in the algal-nutrient relationship. It appears to be a continuous function, with the difference increasing exponentially as the TN/TP ratio decreases. A deviation of the indices at low TN/TP ratios should be expected, since correspondence of the indices is predicated on the algae being phosphorus-limited. If the algae are nitrogen-limited, no correspondence should be expected.

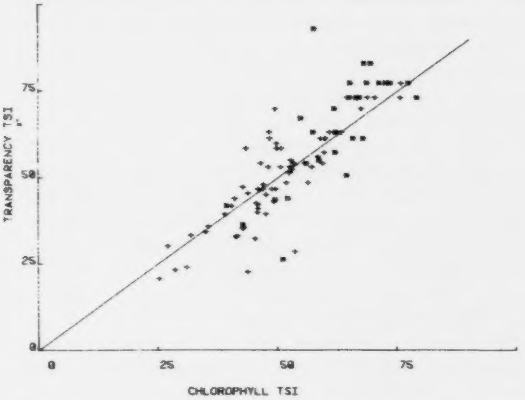

Figure 1. — The relationship between chlorophyll and transparency indices in natural lakes. (+ = TN/TP > 15; * = TN/TP < 15).

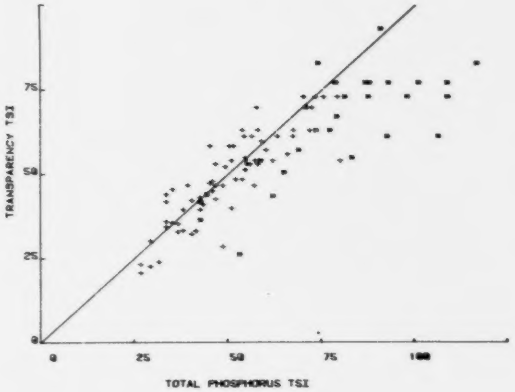

Figure 2. — The relationship between total phosphorus and transparency indices in natural lakes. Symbols as in figure 1.

Impoundments also show this deviation based on the TN/TP ratio, but other overriding factors determine the TSI relationships in these artificial bodies. The chlorophyll-transparency relationship (Figure 5) shows a large number of errant lakes, unlike the relationships found in natural lakes. There is also a poor correspondence between the total phosphorus and chlorophyll indices (Figure 6), again with the most deviation being from those lakes with low TN/TP ratios. However, the total phosphorus-transparency relationship is relatively good, with many of the deviants being lakes with low TN/TP ratios (Figure 7).

The simplest interpretation of the fact that transparency is better related to total phosphorus than

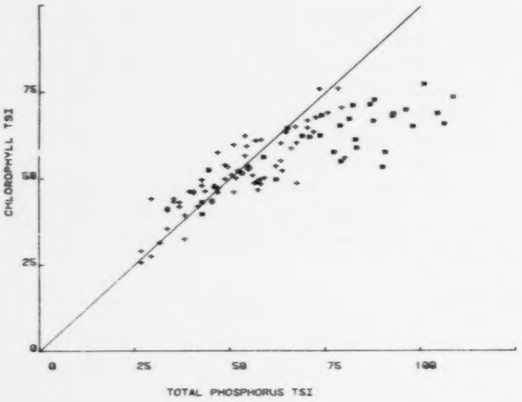

Figure 3. — The relationship between total phosphorus and chlorophyll indices in natural lakes. Symbols as in Figure 1.

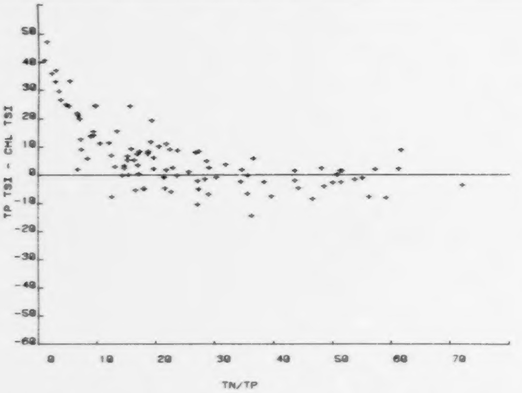

Figure 4. — The deviation of the chlorophyll index from the total phosphorus index as a function of the total nitrogen to total phosphorus (by weight) ratio.

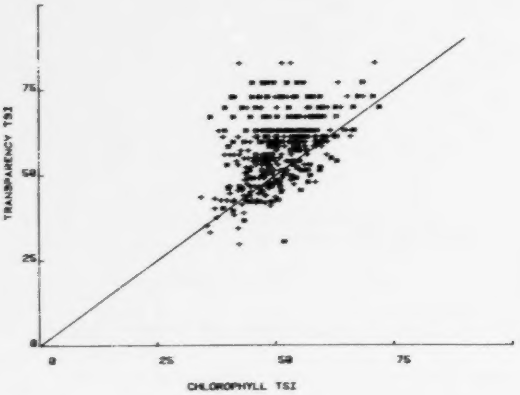

Figure 5. — The relationship between chlorophyll and transparency indices in impoundments. Symbols as in Figure 1.

chlorophyll is that the major attenuator of light in many of the impoundments is a non-algal material, which, however, does contain phosphorus. This non-algal material may be eroded soils or clay. Interpretation requires further information, but again comparing the indices produced evidence that impoundments vary considerably from natural lakes in their nutrient-algal relationships. Perhaps it should be expected that impoundments would be muddier than their natural counterparts, but if this is the case, it would be foolhardy to use predictive models using coefficients derived from natural lakes to predict water quality in impoundments.

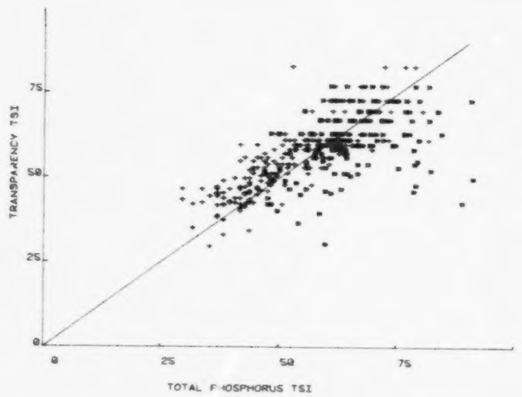

Figure 6. — The relationship between total phosphorus and chlorophyll indices in impoundments. Symbols as in Figure 1.

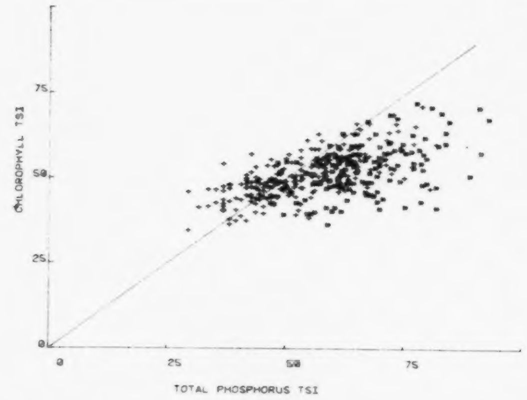

Figure 7. — The relationship between total phosphorus and transparency indices in impoundments. Symbols as in Figure 1.

SEASONAL CHANGES WITHIN LAKES

When adequate seasonal data exist, the three TSI variables within a single lake can be compared. Sometimes, striking deviations in the indices are uncovered. Figure 8 illustrates this in Halsted Bay, Lake Minnetonka, Minn. In each of the 3 years I studied this lake there was a marked decline of algae in late May, at the time the thermocline became established. This type of spring decline in algae has been reported by others, and has been attributed to die-offs of spring species, sinking of the heavier diatoms (Knoechell and Kalff, 1975), and to zooplankton grazing (Fogg, 1975).

The seasonal plot of the indices indicated a marked deviation of the chlorophyll and transparency indices from the phosphorus index in the spring. The increased transparency is certainly the result of decreases in algal chlorophyll, but the amount of phosphorus in the water remains unchanged. Actually, particulate phosphorus decreased but ortho-phosphorus increased. If the algal cells had simply fallen to the bottom, some decrease in total phosphorus might have been expected. The cells must have either lysed while in the epilimnion or have been eaten and the phosphorus excreted (Peters and Rigler, 1973). The coincidence of the year's maximum zooplankton abundance at the time of the decline strongly supports the latter explanation. Other lakes I have examined have also shown this chlorophyll-transparency deviation at the time of high zooplankton densities. Mogadore Reservoir, Ohio was studied by myself and G. D. Cooke in 1976. Two major deviations of the chlorophyll-transparency indices from the total phosphorus index

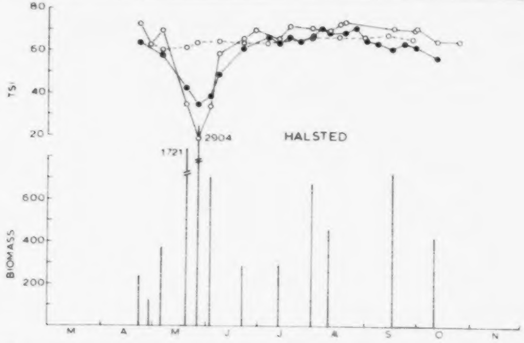

Figure 8. — Upper Graph: The season fluctuations of total phosphorus (O--O), chlorophyll (O—O) and transrency (●—●) indices in Halsted Bay, Lake Minnetonka, Minnesota.
Lower Graph: The dry weight of zooplankton (mg/l) over the same season.

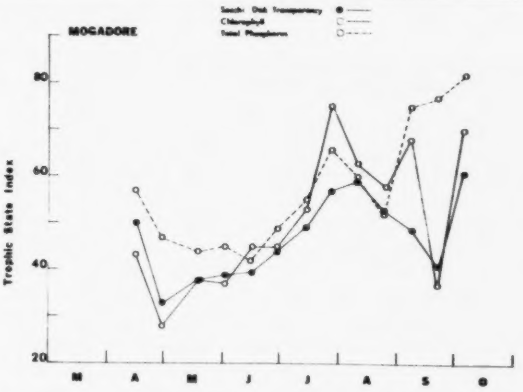

Figure 9. — The seasonal fluctuations of total phosphorus chlorophyll, and transparency indices in Mogadore Reservoir, Ohio.

are seen: one in late April, the other in September (Figure 9). Both periods are characterized by the highest densities of the herbivore *Daphnia galeata*.

Other aspects of Mogadore's index fluctuations are also enlightening. If an index value of 50 is taken to represent the lower limits of eutrophy (Carlson, 1979) then the lake did not exhibit eutrophic levels of algae until mid-July. This lake would have been classified as oligotrophic based on the chlorophyll levels found in May. In Mogadore, these changes are internally driven (Carlson and Cooke, unpubl.), with the rapid rise in phosphorus in late August associated with a major die-off of macrophytes. The wide seasonal variation in trophic state in the reservoir makes using a single trophic designation for a lake questionable. This lake exhibited eutrophic algal characteristics for only 2 to 3 months of its open-water season. The purpose of lake management is better served by a trophic concept and corresponding index that assumes that trophic state is not a constant for a lake but a seasonally changing, dynamic assessment of the lake's condition. Proper management requires measuring the duration of a problem, as well as its extent and cause. Illustrating the seasonal changes in trophic state could have an impact on the assessment of the lake's condition and on the methods used in its management.

CONCLUSIONS

In this paper a trophic state index has been used to identify situations of nitrogen-limitation, non-algal turbidity, and zooplankton-induced algal declines. The index can do this because it provides a set of expected relationships against which data from other lakes can be compared. This method surpasses the simple comparison of raw data because often smaller or regional data sets have internal correlations which may imply relationships that cease to exist when compared against a more global data set. Certainly the idea of what is a normal lake is dictated by the original data set used in the index, but data from some of the world's clearest and worst lakes were included in the original data.

Certainly these relationships could be examined using the original regression relationships rather than the transformed index values. The importance of the transformation is that the comparisons are made in the context of the trophic state concept. The major importance of this concept is that it implies that many aspects of a lake will change as a lake assumes eutrophic characteristics. Even when the index does not measure hypolimnetic oxygen depletion or changes in fish species, the interconnectedness of the lake's biological components is implied. Thus the TSI values take on meaning far greater than do the raw data.

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REGRESSION ANALYSIS OF RESERVOIR WATER QUALITY PARAMETERS WITH DIGITAL SATELLITE REFLECTANCE DATA

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ABSTRACT

Nine Oklahoma reservoirs were sampled monthly from June through October 1979. Water samples were collected concurrent with satellite "passes" and were analyzed for nitrate, nitrite, ammonia and kjeldahl nitrogen, dissolved and total orthophosphate, color, filtrable residue, chlorophylls *a*, *b* and *c*, pheophytin, nephelometric turbidity, and total alkalinity. Additionally, *in situ* data were also gathered concerning temperature, pH, dissolved oxygen, conductivity, wind speed, and Secchi disk extinction depth. Water quality and satellite monitored reflectance data were analyzed utilizing multiple regression techniques. Equations were generated which permit the prediction of chlorophyll *a*, pH, turbidity, color, total alkalinity, and total orthophosphate concentration in Oklahoma reservoirs. Generalization of these relationships to areas outside Oklahoma requires further testing.

INTRODUCTION

Since the launch of the first LANDSAT in July 1972, several studies have evaluated applying satellite based multispectral scanner data to lake or water quality monitoring programs (Bukata, et al. 1974; Rogers, et al. 1975; Bohland, 1976; McKeon, et al. 1977; Scarpace, et al. 1978; Bohland, et al. 1979; Grimshaw and Torrans, 1980).Two of these studies have incorporated concurrently obtained water quality and satellite monitored reflectance data collected from a number of different water bodies on several dates (Scarpace, et al. 1978; Grimshaw and Torrans, 1980). Scarpace, et al. developed regression relationships between trophic class and reflectance values averaged over an entire lake. This study develops regression relationships between LANDSAT monitored reflectance values and specific water quality parameters based upon samples obtained from discrete sampling stations. These relationships will permit the estimation of the concentration of several water quality parameters in reservoirs which were not surface sampled.

SITE SPECIFICITY

To evaluate the use of satellite based multispectral scanner data in water quality monitoring programs it was necessary to compare discrete, site specific water quality data to concurrently obtained site specific reflectance values. Mean reflectance values, obtained by averaging reflectance over the entire water body, were considered inappropriate for use. Consequently, triangulation procedures were used to determine the latitude and longitude of each sampling station. This procedure permitted an accurate comparison of water quality data and satellite monitored reflectance values.

RESULTS

As would be expected, generally more favorable relationships were obtained when the data set was restricted to concurrent satellite and water quality data. Inspection of Table 1 illustrates this point by demonstrating the improvement which can be achieved in the correlation between Band 4 (500 to 600 nm)/Band 5 (600 to 700 nm) ratio and log transformed chlorophyll *a* concentration when only concurrently obtained data are used in the statistical analysis. The extent of this improvement would probably have been even more pronounced had our total data set not been collected very near to the actual satellite coverage dates. Water quality data were collected 1 day before satellite coverage on five occasions; 11 data elements were collected ±11 days or less, and only two entries were off by 18 days. The remaining 22 data elements were collected concurrent with satellite coverage. Table 1 also illustrates the improvement in the Band 4/Band 5 ratio, log chlorophyll *a* correlation coefficient which can be obtained by subjecting the site specific water quality data to a turbidity based cluster analysis prior to correlation with satellite reflectance data.

Figure 2 illustrates this relationship between chlorophyll *a* concentration and LANDSAT multispectral scanner Band 4/Band 5 ratio data. Triangularly shaped symbols represent turbid water samples, defined here to refer to water samples with nephelo-

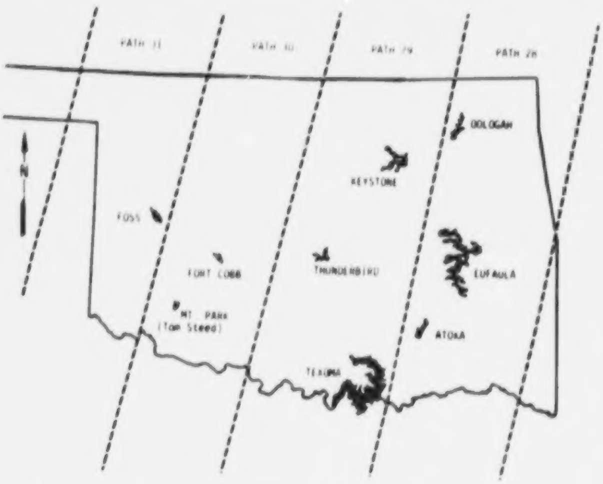

Figure 1. — Reservoirs included in this study.

Table 1. — Correlation coefficients between Band 4/Band 5 ratio and log chlorophyll *a*.

Criterion	Correlation coefficient	Significance level	n
Total data set	0.50142	.01	40
Concurrent data	0.57421	.01	22
Low turbidity concurrent data	0.65726	.01	16

metric turbidity values of 20 N.T.U.'s or greater, while the circular symbols represent clear water samples. Solid symbols indicate that reflectance values were obtained from LANDSAT 3 while open symbols indicate that they were obtained from LANDSAT 2.

A semilog plot of the turbid water Band 4/Band 5 reflectance ratio against chlorophyll *a* concentration approximates a horizontal line, exhibits no statistically significant correlation, and consequently demonstrates the lack of any relationship between these variables. A similar plot (see Figure 2) of the clear water samples, however, indicates that a clear and reasonably strong relationship does exist between these variables, as evidenced by their highly significant correlation (see Table 1).

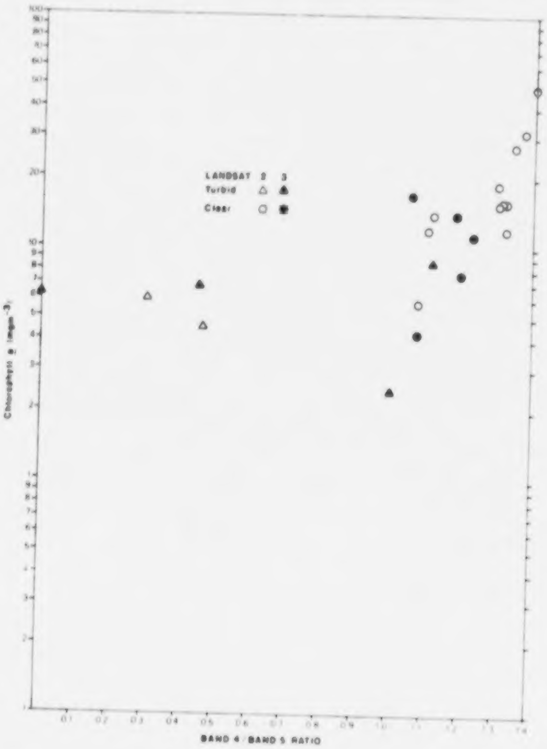

Figure 2. — Semilog plot of chlorophyll *a* concentration and Band 4/Band 5 reflectance ratio.

Further inspection reveals that there is considerable scatter within the data. For example, a reflectance value of 1.05 relates to a chlorophyll *a* concentration of 16.8 mgm^{-3}, while a reflectance value of 1.08 relates to a chlorophyll *a* concentration of 4.4 mgm^{-3}. This relationship is exactly opposite to what one would expect. Because of chlorophyll's absorption and reflective characteristics, a higher Band 4/Band 5 ratio should be related to a higher, rather than a lower, chlorophyll concentration.

These observations suggested that information from more than one band width or band ratio is required to predict chlorophyll *a* concentration with any reasonable degree of accuracy, in spite of the highly significant correlation which had previously been demonstrated between the Band 4/Band 5 ratio and

Table 2. — Regression equations for concentration estimation of Oklahoma reservoir water quality parameters.

Regression equation	Coefficient determination (%)	Significance level	Standard error of estimate
Log chlorophyll *a* = 1.094 + 0.092 (Band 4) - 0.107 (Band 5)	88.8	0.0001	0.42
pH = 9.526 - 0.049 (Band 4) - 0.040 (Band 5) + 0.147 (Band 7)	89.2	0.00o1	0.26
Turbidity = 15.725 - 4.365 (Band 4) + 4.911 (Band 5) - 0.443 (Band 7)	88.7	0.0001	7.92
Color = 32.826 - 4.570 (Band 4) + 4.356 (Band 5)	80.9	0.0001	7.95
Log total alkalinity = 3.877 + 0.033 (Band 4) - 0.031 (Band 5) -0.053 (Band 6) - 1.181 (Band4/Band5)	72.3	0.0001	0.37
Secchi = 0.811 + 0.048 (Band 4) - 0.053 (Band 5)	53.6	0.0001	0.21
Log total ortho-phosphate = -1.022 - 0.064 (Band 4) + 0.058 (Band 5)	41.8	0.0001	0.25

chlorophyll *a*. Consequently, multiple regression analysis was undertaken, utilizing log transformed chlorophyll *a* data as the dependent variable and Bands 4, 5, 6 (700 to 800 nm), 7 (800 to 1,100 nm) and band ratios of 4/5, 4/6 and 5/6 as the independent variable's (see Table 2). The significance of each independent variables contribution to the variance explained was evaluated using a t-test. Variables which did not contribute in a highly significant (.01 level) manner were not used in the equations which were ultimately developed. Results of this analysis are presented in Table 2. Inspection reveals that the multiple regression procedure has successfully explained approximately 89 percent of the variance in the chlorophyll *a* data set.

Regression equations were also subsequently developed to predict pH, turbidity, color, total alkalinity, Secchi disk extinction depth, and total orthophosphate concentration. These equations were developed using the entire data set, while the chlorophyll *a* equation was developed using exclusively concurrent, clear water data. In spite of this fact, several dependent variables, notably pH, turbidity, and color, exhibited very high coefficients of determination.

DISCUSSION

These results suggest the possibility of monitoring several water quality parameters using LANDSAT's multispectral scanner. A more extensive evaluation of the utility of these regression equations, for purposes of trophic classification, will be published in the near future. Initial efforts in this regard, however, are presented in Table 3, where trophic state indices (T.S.I.) (Carlson, 1977) calculated from both observed and predicted chlorophyll *a* concentrations, are tabulated. Graphic analysis of these data (see Figure 3) indicate that the chlorophyll *a* regression equation predicts T.S.I. values with an accuracy of ±7 T.S.I. units. Predictions were most accurate when the chlorophyll *a* concentrations ranged from 13 to about 21 mgm^{-3}. Index estimates obtained when chlorophyll *a* concentrations were from 4 to 11 mgm^{-3} overestimated the T.S.I. by approximately 6 units. Progressively increasing underestimates were obtained when the chlorophyll *a* concentration was in the 32 to 49 mgm^{-3} range.

Table 3. — Observed and predicted chlorophyll *a* concentrations and trophic state indices.

OBSERVED		PREDICTED	
Chlorophyll *a* mgm	T.S.I.	Chlorophyll *a* mgm	T.S.I.
48.8	68.7	22.1	61.0
10.6	53.8	20.6	60.3
14.0	56.5	10.6	53.8
15.8	57.7	18.2	59.1
20.8	60.4	18.1	59.0
7.8	50.8	14.3	56.7
4.3	44.9	8.0	51.0
10.2	53.4	18.5	59.2
5.8	47.8	10.6	53.8
10.2	53.4	18.8	59.4
15.5	57.5	18.1	59.0
18.8	59.4	17.5	58.7
31.8	64.5	21.8	60.8
14.4	56.8	15.2	57.3
13.7	56.3	13.3	56.0
16.8	58.3	7.6	50.5

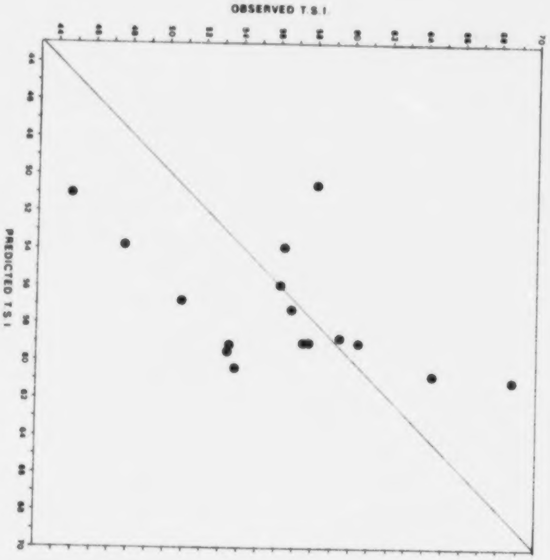

Figure 3. — Comparison of observed and predicted trophic state indices.

CONCLUSIONS

1. Highly significant multiple regression relationships have been demonstrated to exist between several water quality parameters and LANDSAT multispectral scanner data.

2. These equations appear to permit the prediction of chlorophyll based trophic state indices with an accuracy of ±7 T.S.I. units.

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LAKE ASSESSMENT IN PREPARATION FOR A MULTIPHASE RESTORATION TREATMENT

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ABSTRACT

Liberty Lake is a 288 hectare body of water located in eastern Washington. It is a heavily utilized recreational lake when its waters and swimming beaches are not plagued by massive blooms of blue-green algae. In 1974 alum treatment of the lake aimed at late summer and fall release of phosphorus successfully demonstrated the need to control internal cycling of nutrients (especially phosphorus) as well as surface and subsurface input. Macrophytes growing in rich sediments acted as nutrient pumps releasing phosphorus above the floc layer. This event as well as flushing of the bird refuge and marshland to the south of the lake and continued input of septic tanks overcame the alum treatment within 3 years. The 3-year respite was the first in 10 years from blue-green algae problems. Restorative efforts began in 1978-79 with sewering of the lake periphery. Marsh runoff diversion was completed in 1979-80. Suction dredging followed by alum treatment is scheduled for fall 1980. Extensive monitoring of water quality parameters began in late 1977 and has continued to assess each phase of the restoration. The initial results give reason for cautious optimism.

INTRODUCTION

While many of the intricacies of accelerated lake eutrophic processes remain poorly understood, the role and implication of excessive nutrients, especially phosphorus, has been well elucidated (Sawyer, 1947, 1952; Ohle, 1953; Thomas, 1969; Vallentyne, 1974; Edmondson, 1972; Wetzel, 1975).

To deaccelerate, reverse, or at least stabilize the deterioration of a lake's water by overenrichment, the sources of the nutrients must be defined. In addition, the contribution of each source must be determined as accurately as possible and a mechanism set in place to divert, reduce, or mitigate that source of nutrient inflow. Thirdly, unless a concomitant educational program is established, the mitigating efforts may come to naught because a social, economic, or political decision may countermand restorative efforts. Such an action may introduce a new or overload a formerly insignificant nutrient source.

Finally, restorative efforts must be evaluated to predict the future of the lake in question and add to the scientific body of knowledge for lakes of similar background. This paper deals with assessment of multiphase restorative efforts at Liberty Lake, Wash.

STUDY AREA

Liberty Lake (Figure 1) is a softwater lake (288 ha) of glacial origin enclosed on three sides by a small mountain range 300 to 500 meters above the lake surface. Most of the watershed (3,445 ha) lies in this horseshoe-shaped basin, forested by Ponderosa pine,

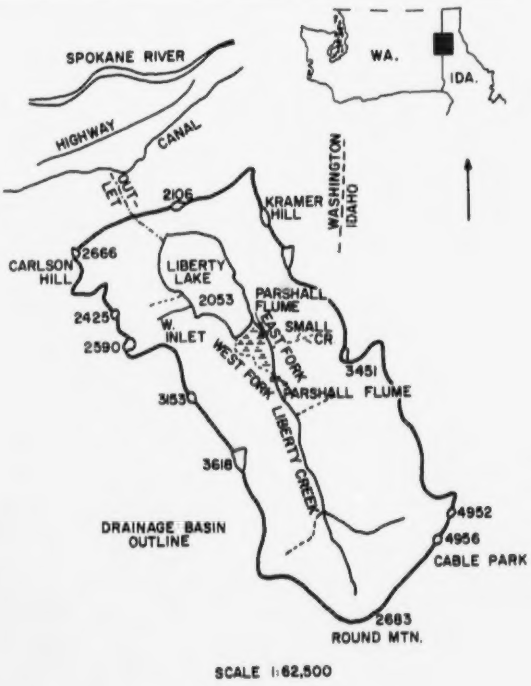


Figure 1. — Liberty Lake and drainage basin.

grand fir, Douglas fir, larch, white pine, and aspen. The major tributary, Liberty Creek, originates in the higher southeastern slopes and passes through Moscow and Springdale soils before reaching the Spokane and Semihoo muck series adjacent to and in a marsh. The stream flows along the eastern margin of the marsh (and until recently, overflowed into it) before entering the lake. Most of the tributary area is underlain by quartz - feldspar - biotite paragneiss. Residential areas occupy 87 percent of the shoreline and overlie relatively shallow soils (Spokane series); gneiss (west side and north shore) and Columbia River basalt (west shore) form the bedrock. A small creek enters the lake from the northwest side. Until 1978-79, waste disposal had been by septic system and an old sewer system built in 1910 which served approximately 40 percent of the residents. In late 1978 and 1979, a collection system was built. It serves about 2,000 permanent residents and diverts 95 percent of the domestic sewage from the lake basin.

The mean residence time of lake waters is 3 years. Approximately 2.76 × 10^6 m^3/yr is lost by seepage, presumably through the bottom at the northern end of the lake. The lake may become weakly stratified for short periods of time during the mid and late summer period.

The lake is heavily used, 80,000 to 100,000 visits per season when the swim areas and beaches are not plagued by massive blue-green algal blooms (*Gloeotrichia, Anabaena,* and *Aphanizomenon*).

A 3-year respite from algal nuisances was made possible by an alum treatment in 1974. Details are described in Funk, et al. (1975) and Gibbons (1979).

ASSESSMENT METHODS

Earlier estimates of stream inflow, lake level, and outflow had been made from Gurley current meters and staff gages in the lake. Precipitation was measured by standard rain and snow collector devices. From these data, Orsborn (1973) developed a water balance. Those estimates were later refined by Copp, et al. (1976) and a nutrient budget developed for the lake by volume weighting flows with phosphorus and nitrogen data collected biweekly during the summer period and monthly during the winter period. During the low flow period of 1977, Parshall flumes were installed on the main stems of Liberty. The flumes were equipped with Manning F 3000 series flowmeters and model S-4040 discrete samplers for continuous flow measurements and water sample collection. Provision was also made with Spokane County Parks personnel for daily inspection and reading of gages in event of equipment failure. Parks personnel were also contracted to read rain gages and evaporation pans. Gurley meters were used to estimate the flow of several small intermittent streams. Storm events and runoff were similarly measured.

Ground Water Inflow

Attempts to measure ground water were made by circumscribing the lake with 17 banks of piezometers as described by McComas (1977).

Sample frequency has been biweekly in the summer and fall and monthly in the winter. Vandalism problems arose when sites were obvious.

Seepage meters (Lee, 1977) were installed in the near shore areas to complement the piezometer banks. Similar vandalism occurred more frequently. Fifteen barrels were set out and only two remained undisturbed; no data were obtained with this method.

Macrophyte Evaluation

Macrophyte growth was estimated by scuba methods along six transects demarked by 100 to 500 m nylon lines laid out on the lake bottom. Plants were collected at 1 m increments of depth using a round metal sampler (.2m^2 area) to delineate the sample area. Three samples were collected at each depth to give a total area sampled of .15m^2. The macrophytes, including roots, were placed in plastic bags, tied, inflated by exhaust air from the diver's tank, and allowed to float to the surface. The boat crew collected and labeled each bag.

In the laboratory, the samples were rinsed to remove sediments, drained, and weighed for wet weight. Subsamples were dried for 24 hours at 100°C. The dried sample was then weighed and ashed at 550°C to obtain ash free dry weight. Additional subsamples were digested to determine nitrogen and phosphorus content.

The data were plotted on a contour map and planimetered to estimate the area of macrophyte growth, weight, and nutrient content.

Macrophyte measurements began each spring at ice off (March or April) and continued until late September or October when at least half of the macrophytes senesce and deteriorate. First estimates were made in fall 1974. Measurements for restoration evaluation began in March 1978 and have continued to date.

Sediment Assessment and Nutrient Release Studies

To assess the contribution of nutrient release from sediments, 10 cores were driven in 1974 by Ewing piston corer. An additional 28 were taken in 1978 by a modified hand-driven piston corer. This latter device used a 12 × 155 cm clear PVC tube as both coring tube and liner. The previously used Ewing corer appeared to force the flocculent sediments away and compacted the upper layers, probably resulting in lower phosphorus values when analyzed. Fourteen cores taken in 1978 were analyzed for nutrient content (phosphorus and nitrogen) and selected metals following methods outlined in the U.S. EPA Laboratory Manual for Bottom Sediments (1969) and Am. Pub. Health Assoc. (1975). Four were subjected to phosphorus release tests. These core samples were augmented by 70 Ekman grab samples taken randomly across the lake to observe the sediment appearance and texture, and to construct a bottom sediment map.

Additional smaller (12 × 33 cm) cores were taken by scuba methods using a small stainless steel piston corer. The small corer could deposit a relatively undisturbed intact core into a laboratory test column.

Both long- and short-term anaerobic, facultative, and aerobic conditions were run on these latter cores to determine release rates of total phosphorus (TP), total soluble phosphorus (TSP), and soluble reactive phosphorus (SRP).

Benthic Invertebrates

Extensive collection of benthic invertebrates began in March 1978. Prior to that time, collections were made on the basis of monitoring the intial alum treatment of the lake. Sampling until August 1978 was carried out at 17 locations and consisted of random Ekman grab samples in each of the areas delineated by the bottom sediment map. For the past 2 years, a modified stratified random sampling method has been adopted which takes into account the three general substrate types (Nelson, 1980). The first can be described as a silt composed of finely divided organic matter, relatively decomposed, and is characteristic of the middle portion and southern end of the lake under 6 to 9 m of water. The second is adjacent to and partially in the littoral zone. This area experiences heavy macrophyte growth during the spring and summer with *Elodea, Potamogeton,* and *Ceratophyllum* being the dominant plants. The sediment is muck-like in appearance and contains partially decomposed plants with some parts readily distinguishable. The third major sediment type is composed of a wide variety of inorganic particle sizes ranging from sand to gravel and rock. Overlying waters are 0 to 6 m in depth.

A random site is selected by superimposing a grid system on a map of the lake. Each grid represents a 60 × 60 m square. The grids within each substrate are numbered and a site selected by random number tables. The number of samples taken from each site depends upon the variation encountered during the previous sample period. The usual number of samples taken is 12. Sample frequency is once every 3 weeks during spring, summer, and fall and once per month in the winter.

Ekman grab samples are used in the open waters and a box type sampler with a bladed screen as described by Minto (1977) is used in areas of heavy macrophyte growth. Material from the samples is initially screened in the field, using three screens in tandem with a high wall on the first. The screen set is composed of numbers 6, 20, and 28 U.S. standard sieve sizes. The screened materials are placed in plastic bottles preserved with 40 percent formalin solution mixed with rose bengal dye (Mason and Yevich, 1967) to facilitate laboratory sorting. Additional screening occurs in the laboratory with U.S. standard sieve sizes 6, 10, 16, 25, 30, and 40. After microscope scanning, the material is placed in a subsampling tray divided into 70, 3 cm^2 units and mixed for random distribution. Subsamples for extensive identification are collected from five of the 3 cm^2 units as determined by a random number table. The data obtained from each subsample are checked for randomness by use of the chi squared test (Elliot, 1971).

Lake Water Quality and Productivity Measurements

Standard field physicochemical measurements were made biweekly during the growing season and monthly in the winter at 1.0 m intervals top to bottom at two lake stations. Measurements included temperature, light transmission, dissolved oxygen, alkalinity, pH, conductivity, and Secchi disk. Both wet chemistry and calibrated probes were used (light - Kahlsico Gemware; conductivity, dissolved oxygen, and pH - Hydrolab II).

Phyto and zooplankton samples as well as samples for laboratory chemistries and chlorophyll *a* were collected by rapid pump methods at the same time, location, and interval as the field measurements. Water for *in situ* carbon-14 productivity measurements were collected in the same manner. Pump manifold and intake funnel were clear PVC; pump line was clear Tygon 1.3 gm (id). Pumping rate was approximately 14 l per minute depending upon depth and was calculated for each sampling date. Zooplankton samples were collected by pumping water through a 60 µm mesh nylon plankton net and cup. Killing and preservation was by formalin, ethanol, and glycerin (Schwoerbel, 1970). Identification and counts were made according to methods outlined in Edmondson (1959), Edmondson and Winberg (1971), Brooks (1957), and Pennak (1978). Successive 1 and 5 l subsamples were examined in the laboratory for abundant and scarce individuals until 50 of the most common individuals were obtained or 20 ml of sample had been analyzed. Generally, phytoplankton counts were made upon subsamples from 1.0 λ unpreserved samples within 24 hours of collection. Preliminary statistical analysis involving the chi squared test (variance to mean ratio) was employed to check for subsample homogeneity.

The remaining portions of the phytoplankton sample were preserved by modified Lugols solution (Schwoerbel, 1970) for additional identification and measurement. If concentration was necessary, centrifugation was employed. Strip count methods were as outlined by Edmondson (1974). Volume measurements of phytoplankton were made as described by Vollenweider, et al. (1974) and Wetzel (1975). Zooplankton biomass ($\mu g/m^3$) was determined using the values of Hall, et al. (1970), Peterka and Knutson (1970), and Bottrell, et al. (1976).

Chlorophyll *a* samples were fixed in the field with $MgCO_3$. Upon return to the laboratory they were immediately filtered through a .45 m Millipore filter and frozen. Chlorophyll *a* was extracted in 90 percent aqueous acetone solution by sonification procedures and measured by reading absorbances on a Beckman model DU 2 spectrophotometer before and after acidification with 1N HCl. Chlorophyll *a* and pheophytin concentration were calculated using formulas contained in Vollenweider (1974) and Am. Pub. Health Assoc. (1975). Calibration of the spectrophotometer was checked periodically by using purified chlorophyll extract (Sigma Chemical Co.). Carbon-14 procedures were carried out *in situ* at both lake stations at 2.0 m intervals. Fifty ml aliquots were filtered through .45 µm Millipore filters and counting was done with a Nuclear Chicago — Mark II Scintillation System.

RESULTS AND DISCUSSION

Water and Nutrient Budget

Restoration of Liberty Lake has been a multiphase program. Preliminary efforts at reducing nutrient flow to the lake began with a temporary diversion structure built in 1978 in cooperation with Spokane County Parks personnel at the bifurcation of Liberty Creek. The diversion allowed a large portion of spring runoff to pass down the West Fork of Liberty Creek without flushing through the marsh to the lake (Figure 2). After the first year of operation it was noted that with capacity stream flow in the East Fork branch nearly 30 percent of the water flow was still being lost to the marsh through breaks in the streambanks and overflow caused by stream bed obstructions. Even with some flushing action occurring, nutrient loading by stream flow was reduced from approximately 148 kg P and 1,137 kg N before diversion to 132.5 P and 831.6 kg N.

A permanent diversion structure was completed in 1979 by M. Kennedy Engineers; the stream channel was also repaired and cleaned. Phosphorus loading was further reduced in 1979 to 117.5 kg but nitrogen was higher (1,191 kg), possibly due to greater interaction between flowing waters and the newly cleaned stream bed. Flow data for 1980 should reflect still lower phosphorus and nitrogen values because of less overflow and flushing. Stream hydrographs and nutrient loading for the East and West Fork inflows are shown in Figures 3 and 4.

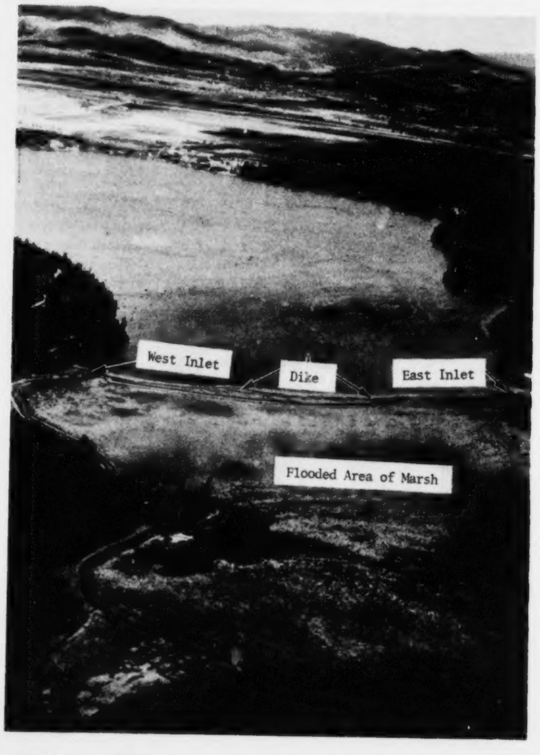


Figure 2. — East and West inlets to Liberty Lake and flooded area of marsh.

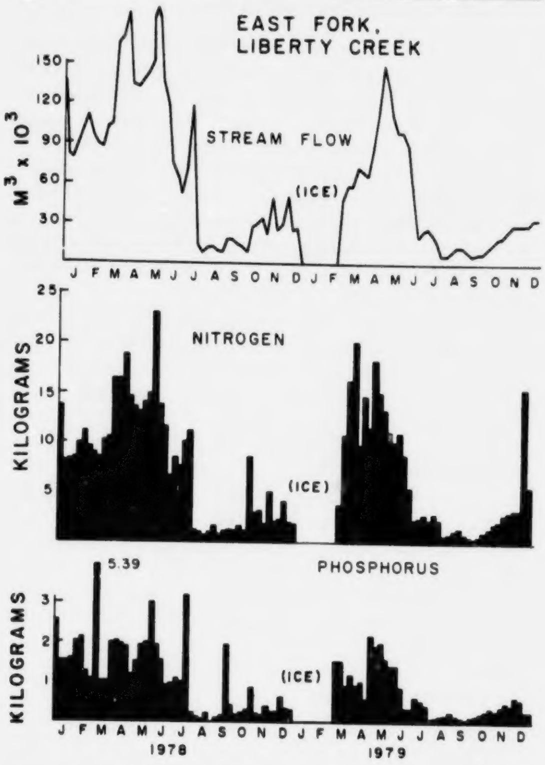


Figure 3. — Hydrograph and nutrient inflow East Fork, Liberty Creek.

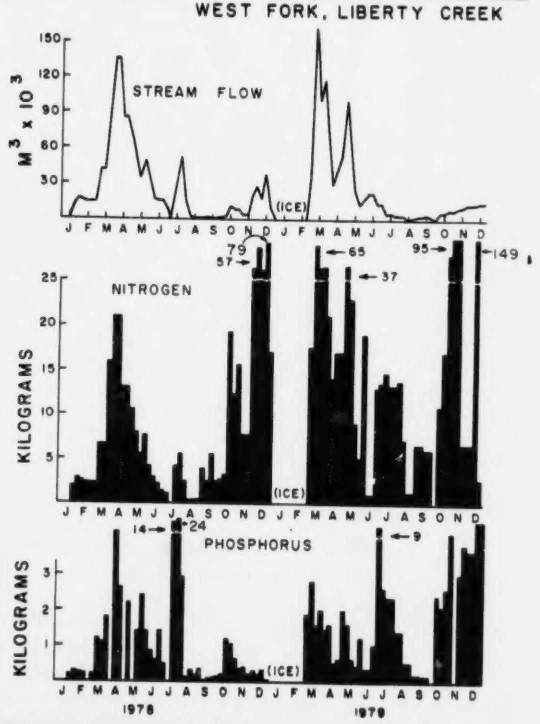


Figure 4. — Hydrograph and nutrient inflow West Fork, Liberty Creek.

Ground Water Measurements

Preliminary estimates of groundwater input made prior to completion of the sewer collection system indicated that the immediate area (784 ha) around the lake contributed approximately 6.3 × $10^5 m^3$ of water which carried approximately 150 kg P and 1,717 kg N toward the lake (Rector, 1979). The remaining portion of the watershed is relatively undisturbed (2,660 ha) and its nutrient addition is believed to be minor in comparison, contributing only about 12 kg as P and 57 kg as N to the lake. Table 1 shows mean values of piezometer samples for 1978-79.

It is interesting to note that at Dreamwood Bay (site 8), a number of homes have been built on fill across the natural drainage. After the collection system was completed in 1979, the piezometer samples show sharply reduced nutrient values. It is expected that most other areas will not show such immediate improvement because of prolonged drainage bed leaching and perched water table contribution. These later phenomena were graphically demonstrated when trenches were opened for the collection system in 1979. Percolation had occurred in the sandy loam soils until a plugging of soil pores occurred or a clay lens prevented downward movement. The effluent then ponded and moved to the surface or laterally until it reached the soil surface on the downgrade.

Considerable groundwater outflow from the lake occurs in the northern portion of the Lake; Orsborn (1973) estimated the flow to be 2.76 × 10^6 m^3/yr. He used existing well logs to show a sand and gravel aquifer 6 m in thickness and approximately 1.5 km wide. One well (25/145-16 Cl) is centrally located in the aquifer and its elevation closely follows that of the lake. We estimate a loss of 40 to 50 kg of P/yr from the lake bottom to the sand gravel aquifer based upon analysis of the well water and flow.

Table 1. — Mean groundwater concentration of phosphorus and nitrogen at Liberty Lake.

Drainage Basin	Piezometer Sample Site	Total Phosphorus	Total Nitrogen
Northwest Side	5,12, 13, 14	.26	2.54
North End	22, 23, 24, 25 + Northwest side values	.153	1.41
East Side	3	.26	1.78
MacKenzie Bay	6, 7	.065	.54
County Park	9, 10	.036	.44
Dreamwood Bay	8	.25	5.72
Main Watershed	1, 2, 28, 29, 30	.039	.008

Sediment Contribution

Earlier investigations (Funk, et al. 1976, 1979) had shown that much of the deposited material in Liberty Lake such as the gravel, clay, and sand release relatively few nutrients. Recent intensive laboratory work (Mawson, 1980) has verified that release of nutrients from sediments located in the southern portion of the lake could in conjunction with macrophyte decline account for huge algal populations. Two sediment types were tested, an organic refractory silt (ROS) representing about 70 ha and a heavy organic muck (HOM) making up 68 ha. In one test series oxygen was added to waters overlying the sediment column (aerobic). In another series of tests (faculative) the columns were open at the top and atmospheric oxygen was allowed to equilibrate with column water. Finally, in an anaerobic series oxygen was removed by adding sulfide. Conditions during tests are shown in Table 2. Summarized results for the ROS sediments are shown in Table 3 and results for the HOM sediment type are shown in Table 4.

Table 2. — Average dissolved oxygen concentration, average pH and standard deviations for HOM and ROS (Mawson, 1980).

	DO (mg/l)	Standard Deviation	pH	Standard Deviation
		HOM		
Anaerobic	0.0	---	6.65	0.31
Facultative	2.71	0.41	6.63	0.27
Aerobic	8.36	0.95	6.97	0.46
		ROS		
Anaerobic	0.0	---	6.55	0.16
Facultative	3.1	0.53	6.70	0.15
Aerobic	6.84	0.96	7.02	0.38

Table 3. — Summary of number of observations (n), slope of concentration over time (k), correlation coefficients (r), release rates (k*), and confidence levels for average observed concentrations for ROS sediment (Mawson, 1980).

	N	K(mg/l-day)	r	k*(μg/m^2-hr)	Confidence levels	Analysis
Anaerobic	28	0.007	0.909	12.7	99%	T-P
Facultative	30	0.001	0.330	1.22	<99%	T-P
Aerobic	16	0.000	0.092	0.186	<95%	T-P
Anaerobic	34	0.001	0.772	7.10	99%	TSP
Facultative	26	0.000	0.455	1.40	95%	TSP
Aerobic	17	0.000	0.319	0.3057	<95%	TSP
Anaerobic	34	0.001	0.807	2.75	99%	SRP
Facultative	26	0.000	0.277	0.393	<95%	SRP
Aerobic	17	5.283×10^{-8}	0.000	9.79×10^{-5}	<95%	SRP

Table 4. — Summary of number of observations (n), slope of concentration over time (k), correlation coefficients (r), release rates (k*), and confidence levels for average observed concentrations for HOM sediment (Mawson, 1980).

	N	K(mg/l-day)	r	k*(μg/m^2-hr)	Confidence levels	Analysis
Anaerobic	34	0.003	0.693	5.42	99%	TP
Facultative	82	0.001	0.839	2.48	99%	TP
Aerobic	26	0.000	0.411	0.399	95%	TP
Anaerobic	34	0.002	0.707	2.698	99%	TSP
Facultative	82	0.001	0.779	1.41	99%	TSP
Aerobic	26	6.9×10^{-6}	0.026	0.011	<95%	TSP
Anaerobic	34	0.002	0.832	3.071	99%	SRP
Facultative	82	0.001	0.750	1.10	99%	SRP
Aerobic	34	1.23×10^{-5}	0.290	1.87×10^{-2}	<95%	SRP

As expected, maximum release rates occurred under anaerobic conditions for the phosphorus component measurements. Release rates from ROS sediments were also much greater, almost by a factor of 2, than the HOM sediments (with the exception of SRP).

Mawson (1980) believes that this may be due to more interstitial phosphorus present in the HOM as well as greater biological activity. The release of phosphorus from the ROS appears to follow a diffusion pattern while that of the HOM is much more complex. Bottom sediment types are shown in Figure 5.

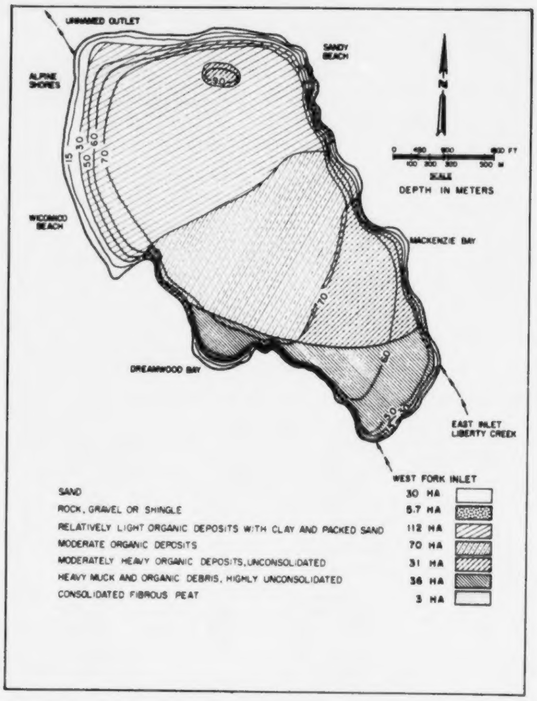
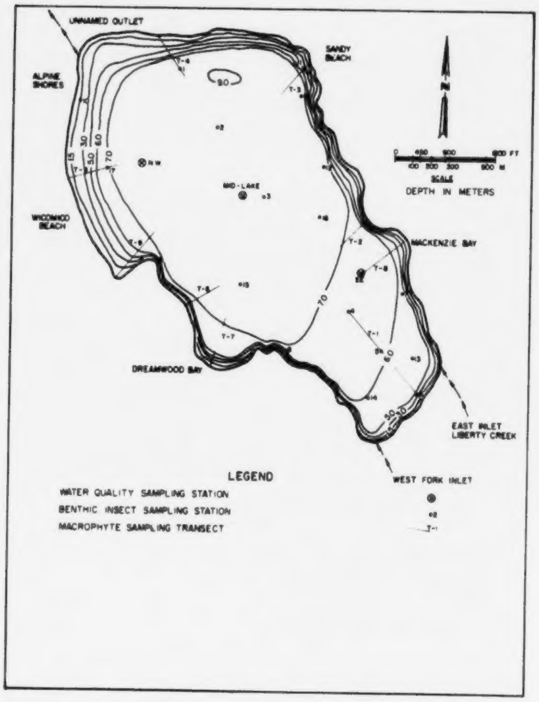

UNNAMED OUTLET
SANDY BEACH
ALPINE SHORES
WICOMICO BEACH
MACKENZIE BAY
DREAMWOOD BAY
EAST INLET LIBERTY CREEK
WEST FORK INLET
N
0 450 900 1800 FT
100 200 300 500 M
SCALE
DEPTH IN METERS
15
30
50
60
70
90
SAND 30 HA ☐
ROCK, GRAVEL OR SHINGLE 5.7 HA ▩
RELATIVELY LIGHT ORGANIC DEPOSITS WITH CLAY AND PACKED SAND 112 HA ▨
MODERATE ORGANIC DEPOSITS 70 HA ▨
MODERATELY HEAVY ORGANIC DEPOSITS, UNCONSOLIDATED 31 HA ▨
HEAVY MUCK AND ORGANIC DEBRIS, HIGHLY UNCONSOLIDATED 36 HA ▨
CONSOLIDATED FIBROUS PEAT 3 HA ☐

Figure 5. — Bottom sediment characteristics.

Macrophyte

Macrophytes were collected along a series of transects as shown in Figure 6. First measurements were made in 1974 and at that time approximately 58,000 kg (dry weight) of plants covered 80 ha of bottom area. Approximately half of the aquatic plants die in the fall. In 1977 at the time of fall dieback a moderately heavy algal bloom of *Anabaena flos-aquae* immediately developed. This increased shading of viable plants and stimulated production of blue-green aglal cellular products, causing additional macrophyte deterioration. These events coupled with drought conditions and unseasonably high water temperatures increased blue-green algal growth, leading to additional shading and byproducts. Ultimately, over 75 percent of the macrophyte beds deteriorated and floated to the surface in large rafts. By the onset of the present investigation in 1978, maximum standing crop was about 60 percent of earlier years and the previously dominant macrophyte *Ceratophyllum demersum* had been replaced by *Elodea canadensis*, especially in the southern portion of the lake. As noted by Figure 7 and a mean of 3 years of data, the most prolific growth is in the richer, more unconsolidated sediments of the southern end. Since this area has been proposed for dredging (to remove the top 0.5 meter of rich sediments) a considerable amount of prime macrophyte bed area will also be removed.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SCALE
DEPTH IN METERS
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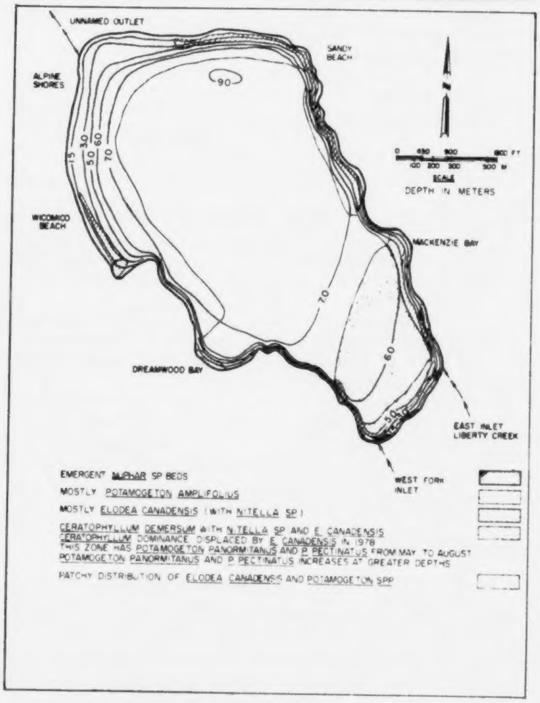
LEGEND
WATER QUALITY SAMPLING STATION ●
BENTHIC INSECT SAMPLING STATION
MACROPHYTE SAMPLING TRANSECT T-1

Figure 6. — Sampling stations and macrophyte collection transects.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0 450 900 1800 FT
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SCALE
DEPTH IN METERS
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EMERGENT NUPHAR SP BEDS
MOSTLY POTAMOGETON AMPLIFOLIUS
MOSTLY ELODEA CANADENSIS (WITH NITELLA SP)
CERATOPHYLLUM DEMERSUM WITH NITELLA SP AND E. CANADENSIS
CERATOPHYLLUM DOMINANCE DISPLACED BY E. CANADENSIS IN 1978
THIS ZONE HAS POTAMOGETON PANORMITANUS AND P. PECTINATUS FROM MAY TO AUGUST
POTAMOGETON PANORMITANUS AND P. PECTINATUS INCREASES AT GREATER DEPTHS
PATCHY DISTRIBUTION OF ELODEA CANADENSIS AND POTAMOGETON SPP

Figure 7. — Distribution of aquatic macrophyte species.

Table 5. — Macrophyte biomass during maximum standing crop, August 2, 1978.

Location	Depth (m)	Area (m^2)	Ash free dry weight per area (g/m^2)	Total org. mass (kg)	% P/ contour	Total P/contour (kg)
Southern end and along east side of lake including MacKenzie Bay	0-1	14.6×10^3	31.7	463	.20	.93
	1-2	18.8×10^3	126.0	2,370	.21	4.98
	2-3	38.6×10^3	171.0	6,600	.21	13.86
	3-4	132.0×10^3	98.5	13,000	.16	20.80
	4-5	176.0×10^3	41.9	7,390	.13	9.61
	5-6	86.6×10^3	13.0	1,130	.13	1.47
	Sub-total southern portion of lake			31,000		51.65
Eastern portion of lake from MacKenzie Bay north to public launch area	2-4	36.5×10^3	38.9	1,420	.19	2.70
Dreamwood Bay	3-4	5.22×10^3	20.5	107	.18	.19
Off southern end of Wicomico Beach	2-3.5	13.6×10^3	171.0	2,320	.19	4.41
	Total lake			35,000		58.95

Table 6. — Macrophyte biomass during maximum standing crop, July 17, 1979.

Location	Depth (m)	Area (m^2)	Ash free dry weight per area (g/m^2)	Total org. mass (kg)	% P/ contour	Total P/contour (kg)
Southern end and along east side of lake Including MacKenzie Bay	0-1	14.6×10^3	104	760	.15	1.14
	1-2	18.8×10^3	89	2,670	.20	5.34
	2-3	38.6×10^3	101	5,021	.39	19.58
	3-4	132.0×10^3	121	17,491	.23	40.23
	4-5	176.0×10^3	169	14,465	.38	54.97
	5-6	86.6×10^3	114	2,338	.49	11.46
	6-7	8.7×10^3	116	433	.36	1.56
	Sub-total southern portion of lake			43,178		134.28
Eastern portion of lake from MacKenzie Bay north to public launch area	2-4	36.5×10^3	120	5,291	.12	6.35
Launch area to Wicomico Beach	2	13.6×10^3	184	4,243	.23	9.76
	Total lake			57,712		150.39

Table 7. — Dominant macrobenthic fauna of Liberty Lake. (Preliminary list by class, order or family where possible.)

Family Chironomidae
- Ablasbesmyria
- Chironomous
- Cryptocladopelma
- Cryptochironomous
- Endochironomous
- Glyptodendipes
- Polypodium
- Procladius
- Pseudochironomous

Family Chaoboridae
- Chaoborus

Family Ceraptopogonidae
- Palpomyia
- Alloaodomynia

(In addition, individuals of the classes Pelecypoda, Gastropoda and Oligochaeta are present in moderate to high numbers.)

Order-Ephemeroptera

Order-Odonata

Benthic Invertebrate Assessment

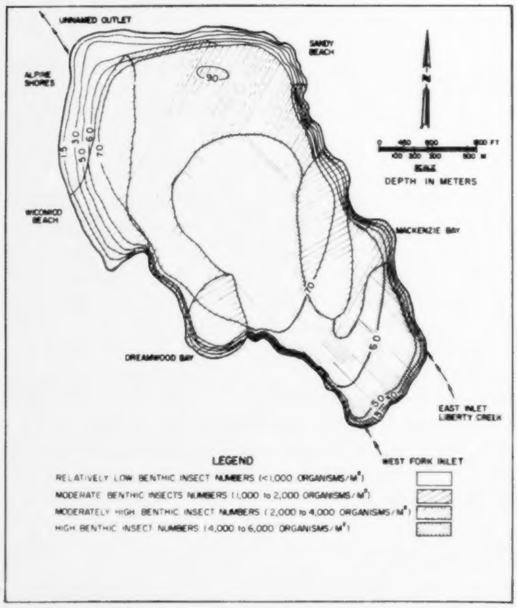
To assess effects of restoration efforts upon higher aquatic food chains, extensive collections of benthic invertebrates began in March 1978. Attempts are being made to classify the organisms to genera and to species where possible. Preliminary results indicate that higher numbers of organisms are found in the mid-lake and southern end of the lake. Large chironomids were found in abundance in the mid-lake sediments which intergrade between the heavy organic deposits characteristic of the southern end and the lighter organic material found in the northern end of the lake.

Dominant benthic forms found in the soft silty sediments were dipteran larvae of the genera *Chironomus, Procladius,* and *Chaoborus*. *Tanytarsus* was also present in fewer numbers. *Chironomus* sp were also found in numbers of 1,700 to 2,500 organisms/m² in the northern and mid-lake sections as well as 430 to 1,460 organisms/m² in the heavier silt and organic debris at the southern end. Figure 8 indicates the distribution found to date. Table 7 lists the dominant forms.

Nelson (1980) will be publishing pre- and post-treatment results. Our earlier studies indicate that with an alum treatment benthic invertebrates appeared to increase moderately for a short period of time before returning to previous population levels. Narf (1978) also indicates that this may be the case, as well, in alum-treated Wisconsin lakes. As only a portion of Liberty Lake will be suction dredged in the fall of 1980, it will be possible to study the effects of dredged and undredged areas in conjunction with alum treatment.

In other field studies concerning benthic organisms, over 100 fish stomach samples were collected during the spring and summer of 1978 with the aid of fishermen and the homeowners association. Much of the data appears to be invalid because of the intensive put and take fisheries. Most of the stomachs contained corn, artificial eggs, hamburger bits, and pull tabs rather than benthic forms. Two permit requests made by our fisheries biologist to take fish by shocking or netting were denied by the Washington State Department of Game.

In laboratory studies being completed now (Lamb, 1980) a chironomid, *Tanytarsus dissimilus*, is being subjected to acute and long-term alum toxicity tests. At this time 240, 320, 560, and 750 mg/l of alum have not shown acute toxicity to the organisms. In tests to date all individuals appeared to remain fairly active and have had no difficulty moving through the floc layer (1.1 mm to 3.5 mm in depth) and actively injesting algal cells, *Selenastrum capricornutum*, supplied for food. It is expected that these data will be reviewed and submitted for publication shortly.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SCALE
DEPTH IN METERS
WICOMICO BEACH
MACKENZIE BAY
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DREAMWOOD BAY
50
30
EAST INLET LIBERTY CREEK
LEGEND
WEST FORK INLET
RELATIVELY LOW BENTHIC INSECT NUMBERS (<1,000 ORGANISMS/M²)
MODERATE BENTHIC INSECTS NUMBERS (1,000 to 2,000 ORGANISMS/M²)
MODERATELY HIGH BENTHIC INSECT NUMBERS (2,000 to 4,000 ORGANISMS/M²)
HIGH BENTHIC INSECT NUMBERS (4,000 to 6,000 ORGANISMS/M²)

Figure 8. — Benthic invertebrate distribution.

Lake Water Quality and Productivity Measurements

Physicochemical Conditions

Generally the lake exhibits weak thermal stratification during the summer with loss of oxygen in the hypolimnion and occasional anoxic conditions (Gibbons, 1976). During this investigation the lake did not stratify but oxygen levels were reduced to below 2.0 mg/l and eventually to zero near the sediments by August. For purposes of brevity Table 8 lists only maximum, minimum, and mean measurements taken to date. Extensive measurements taken at 1.0 m intervals are on computer file at WSU and in LEI data. The concentration of phosphorus versus time has revealed no discernible seasonal pattern. The usually low concentration of phosphorus is probably due to its almost instantaneous uptake by algae and macrophytes. Nitrogen to phosphorus ratio was 17:1 and Schindler (1978) suggests that when N:P is more than 10:1 phosphorus is the most limiting of the two. As Wetzel (1975) has noted, the concentrations of SRP and TSP are not as significant as the rate of interchange between SRP and TSP and particulate phosphorus in the water. The ability of the blue-greens to accumulate phosphorus far in excess of their immediate needs (Fogg, et al. 1973; Whitton, 1973), helps to account for our repeated observation that masses of *Anabaena flos-aquae*, *A. spiroides*, and *Gloeotrichia echinulata* first appear in the vicinity of decaying macrophytes and near the bottom sediments before rising in the water column.

Table 8. — Summary of 1978-79 water quality conditions at Liberty Lake (µg/l except where noted).

	Southeast			Northwest		
Parameter	Mean	Min.	Max.	Mean	Min.	Max
TP	30.0	10.0	75.0	3.5	8.0	78.0
TSP	7.5	2.5	17.5	7.5	2.6	12.5
SRP	1.5	1.0	5.2	4.0	1.0	5.6
N-Ammonia	10.0	5.0	60.0	10.0	6.0	70.0
N-Nitrite-Nitrate	15.0	10.0	78.0	20.0	10.0	58.0
Total N	400.0	275.0	550.0	360.0	320.0	545.0
Alkalinity (mg/l)						
HCO_3	21	13.0	36.0	20.0	13.0	36.0
CO_3	<1.0	0.0	3.0	<1.0	0.0	3.0
CO_2	2.0	0.0	7.0	2.0	0.0	17.0
D.O. (mg/l)	10.0	8.0	16.0	8.0	0.0	16.0
pH (-log H^+)		6.5	8.5		6.0	8.6
Secchi Disk (M)	3.3	1.8	6.0	3.6	1.8	6.0
Temperature	18.2	0.0	25.0	18.2	0.0	25.0
Chlorophyll *a*	8.0	1.0	40.0	10.0	1.0	25.0

Phytoplankton Productivity

The phytoplankton of Liberty Lake produced approximately 8.6×10^5 kg of organic carbon in 1978 and 5.9×10^5 kg of organic carbon in 1979. The annual rate of productivity was estimated at 300 g C m²/yr in 1978 and 205 g C m² in 1979. We would like to attribute the reduced productivity to the first restoration measures, the completion of the sewage collection system and the diversion of spring runoff waters from flooding the marsh. However, part of the reduced phytoplankton productivity is most likely due to competition from increased macrophyte growth as can be noted by comparing Tables 5 and 6; cooler weather and changed patterns of precipitation are also other factors. Gibbons (1980) has observed that diatoms made up over 50 percent of the standing crop during the same period. He also noted that according to Hutchinson (1967) three of the four prominent species (*Fragilaria crotonensis, Melosira granulata,* and *Tabellaria fenestrata*) are indicative of eutrophic waters. Figure 9 shows primary productivity for 1978-79. Figures 10, 11, 12, and 13 show mean cell volumes of dominant species of blue-greens and diatoms measured over the same period. *Ceratium hirundinella*, a relative newcomer and prominent contributor to lake phytoplankton populations, is also shown.

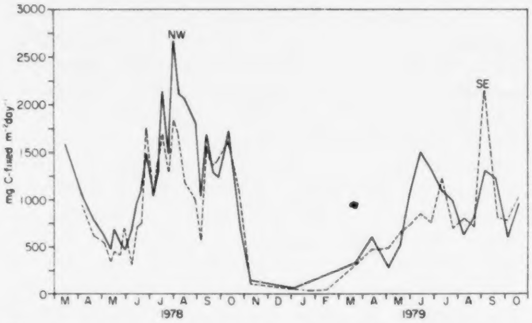

Figure 9. — A monthly summary of carbon 14 productivity (integrated) for Liberty Lake Stations.

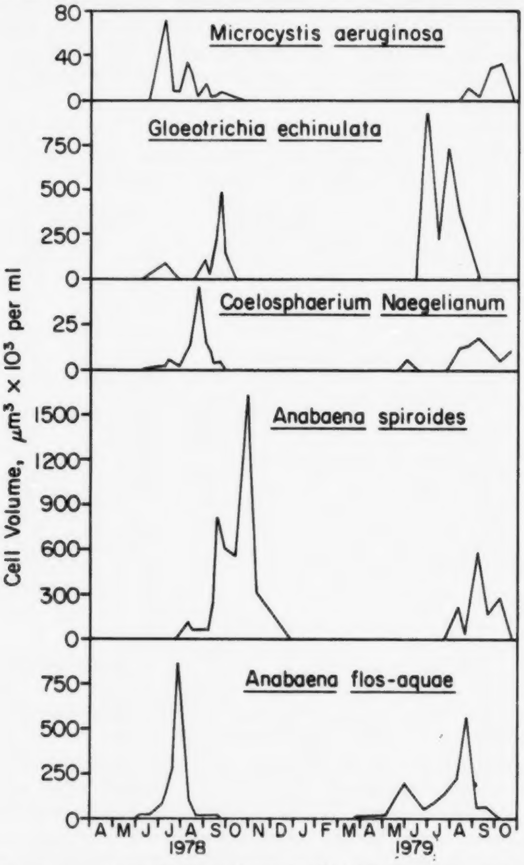

Figure 10. — Contribution to biomass of blue green algae at the Southeast Liberty Lake Station.

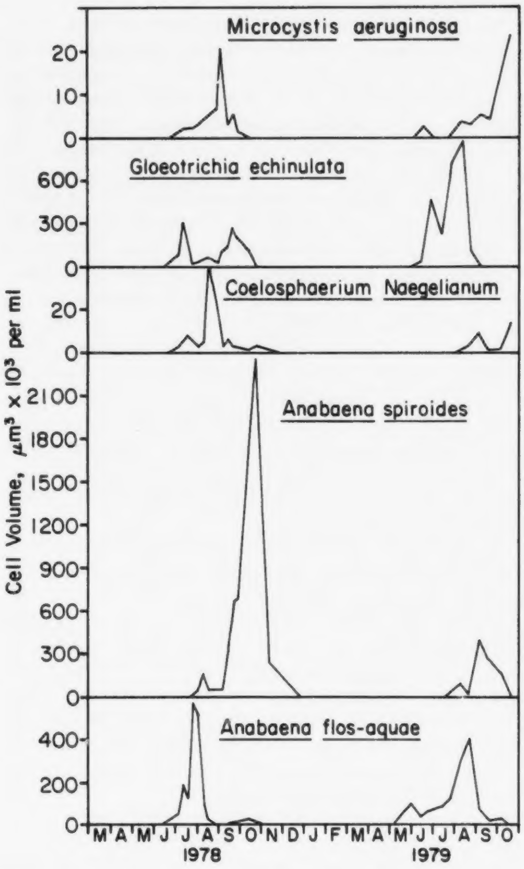

Figure 11. — Contribution of biomass by blue-green algae at the Northwest Liberty Lake Station.

Chlorophyll *a*

Chlorophyll *a* has shown a sharp reduction over the past 2 years; in 1977 lake values exceeded 30 μg/l for a 2-week period and ranged as high as 240 μg/l. This occurred after the early decline of macrophytes and subsequent rise of *Anabaena flos-quae, A. spiroides, Gloeotrichia echinulata* blooms (Figure 14). *Aphanizomenon flos-aquae* appeared for a short period under the ice the first week of January 1979 after a late die-off of some of the remaining macrophytes. Chlorophyll *a* levels generally ranged from 2 to 20 μg/l at the southeast station (Figure 15). Chlorophyll *a* did reach 40 μg/l at one level at the southeast station in late October 1979.

Zooplankton

Thirty-four species and 28 genera of zooplankton have been recognized by Gibbons (1980) at Liberty Lake during the 1978-1979 study years. Eleven species were Rotifera, 17 were Cladocera, five were Eucopepoda, and one was a Diptera. Rotifera dominated the community by numbers (87 percent) but made up only 4.3 percent of the biomass. The grazing Cladocera made up 5 percent of the numbers but accounted for 58 to 67 percent of the biomass. The Calanoida contributed 9 percent of the abundance but 25 percent of the biomass over the study period. The contributions made by the major species are shown in Table 9. The careful documentation of the species composition and numbers should aid in assessing the effects of restoration measures upon secondary production.

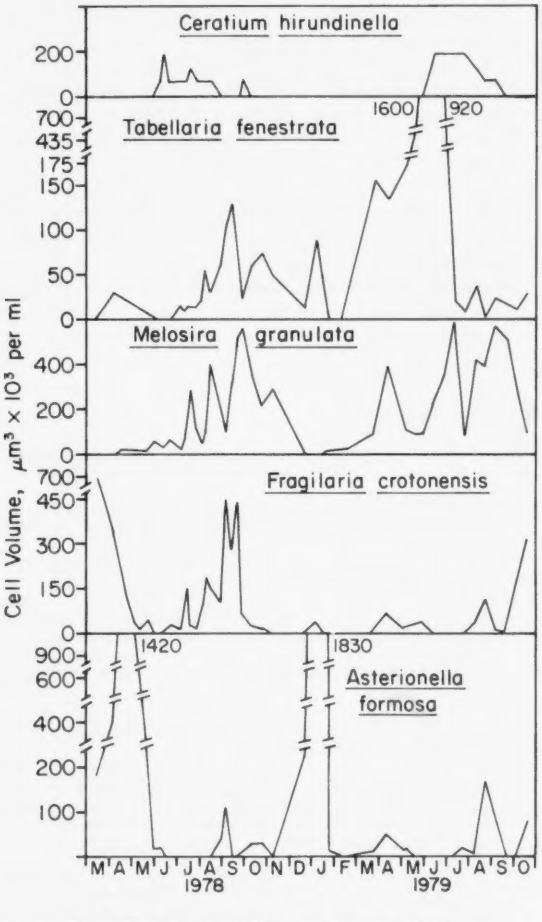

Figure 12. — Contribution to biomass by phytoplankton other than blue-green algae at the Southeast Liberty Lake Station.

SUMMARY

The low buffering capacity, shallow depth, and relatively long-term detention time of Liberty Lake waters has precluded a large one-time monophase restoration effort. In addition, the high usage rate by residents, park visitors, fishermen, and water sports enthusiasts has made a stepwise treatment the most judicious route to follow. This procedure also insures the protection of commerical interests whose livelihood depends upon seasonal use of the lake.

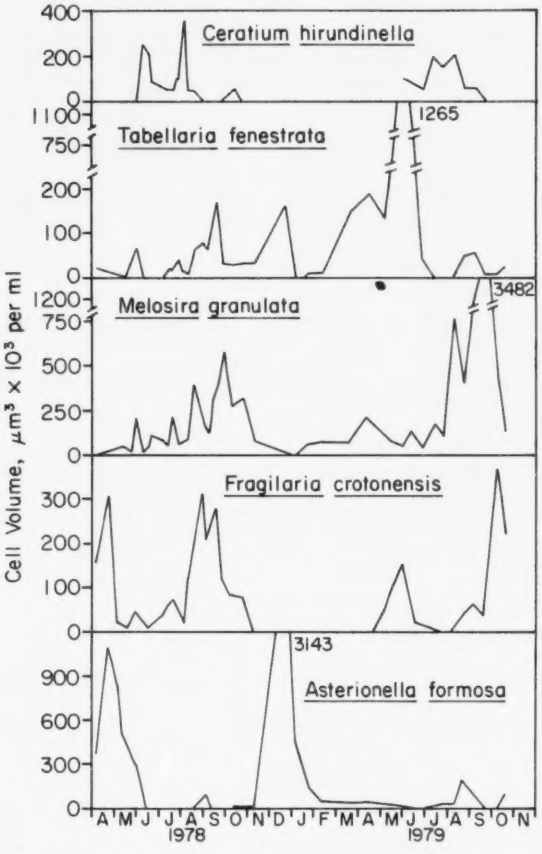

Figure 13. — Contribution to biomass by phytoplankto other than blue-green algae at the Northwest Liberty Lake Station.

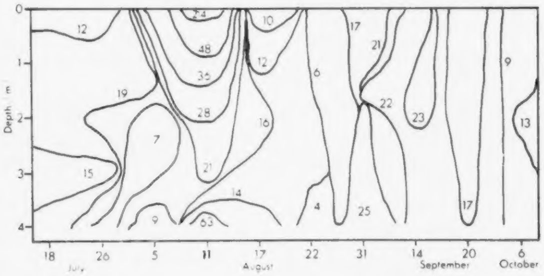

Figure 14. — Southeast Liberty Lake chlorophyll *a* measurements (ug/l) July — October, 1977.

The most distinctive asset in terms of restoration protection has been the undisturbed upper watershed of Liberty Lake. Phosphorus and nitrogen content of inflowing waters is very low unless overflow flushes additional nutrients from the adjoining marsh.

A 15 to 20 percent reduction in phosphorus input has been achieved by diverting flood waters away from the marsh and to the lake through repair of the West Fork of Liberty Creek.

Sewering 95 percent of the residential area around the lake has diverted another 150 to 170 kg of nitrogen

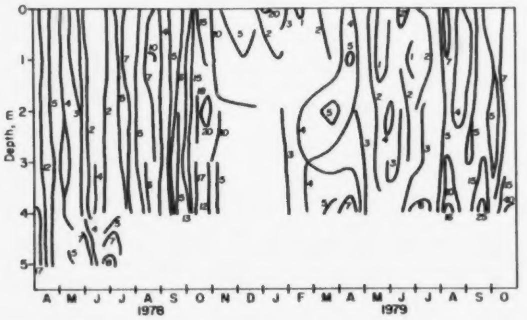

Figure 15. — Chlorophyll *a* measurements at Liberty Lake Southeast Station (ug/l).

Table 9. — Summary of principal taxa comprising the zooplankton community of Liberty Lake, Wash. (9/27/78-8/22/79), including percent composition in terms of density (#'s/m³) and dry weight standing crop ($\mu g/m^3$) over all dates at the northwest and southeast stations (Gibbons, 1980).

Taxon	Percent Composition (#'s/m3) NW	Percent Composition (#'s/m3) SE	+g/m3 NW	+g/m3 SE
Cladocera	4.1	2.8	66.5	57.7
Bosmina longirostris	*	*	0.1	0.2
Chydorus sphaericus	0.5	0.6	1.9	3.2
Ceridaphnia lacustris	*	*	0.1	0.1
Diaphanosoma brachyurum	0.1	0.3	0.6	3.0
Daphnia pulex	0.5	0.3	18.2	13.9
Daphnia schodleri	0.3	*	9.4	2.6
Daphnia galeata mendotae	1.3	0.9	19.7	21.4
Daphnia thorata	0.8	0.4	12.0	8.9
Daphnia immatures	0.5	0.3	4.2	3.9
Leptodora kindtii	*	*	0.1	0.2
Rotifera	86.3	87.2	4.3	8.0
Keratella cochlearis	24.8	41.0	1.7	4.6
Kellicottia longispina	38.3	28.2	0.7	0.9
Trichocera cylindrica	2.9	1.8	*	*
Gastropus sp.	0.6	1.0	*	*
Conochilus sp.	0.8	0.2	*	*
Polyarthra spp.	18.8	14.8	1.8	2.2
Calanoid Eucopepoda	9.3	9.8	25.5	30.1
Nauplii	7.2	8.3	2.0	3.8
Copepodids	1.6	1.1	8.6	10.0
Diaptomus reighardi	0.5	0.4	14.9	16.3
Cyclopoid Eucopepoda	0.3	0.2	2.9	3.5
Copepodids	0.3	0.2	1.4	1.1
Mesocyclops Leuckarti	*	*	0.2	0.4
Macrocyclops albidus	*	*	1.3	1.8
Diptera	*	*	0.8	0.7
Chaoborus sp.	*	*	0.8	0.7

* ' *0.1%

from its yearly movement to the lake. It is estimated that septic tank drainage beds will continue to leach for another 4 to 7 years depending upon the hydraulic pressure placed on them.

The importance of reducing the macrophyte beds and their rich substrata cannot be overemphasized. Mawson (1980) has shown that when the sediments become anaerobic a potential 150 to 280 kg phosphorus could be released. Based upon Mackenthun and Ingram's (1967) estimates of phosphorus contained in algae, the potential release of phosphorus from macrophytes and sediments theoretically produces 77 to 100 metric tons of algae in the water. After dieoff and decay of macrophyte beds and subsequent algal blooms in 1971 and 1973, we estimated approximately 60 to 70 metric tons of debris on the beaches alone (Funk, et al. 1975).

Suction dredging is planned for fall 1980 to remove up to 33 percent of the rich top sediment and about 50 percent of the heavier macrophyte growth area. Another alum treatment of 10 mg/l is planned to coincide with the dredging to reduce phosphorus released by suspended sediments. This latter treatment will also help seal freshly exposed sediments and aid in breaking the nutrient cycle without overwhelming the buffering capacity of the lake.

A program is being planned with M. Kennedy Engineers to reduce urban runoff both mechanically and by educating the residents on the importance of minimum lawn fertilization, litter control, and clean streets.

The ultimate goal is to reduce controllable nutrient input (especially phosphorus) by about 40 percent and to assess as accurately as possible the value of each restoration measure put into practice.

Barring undesirable land use practices beyond controlled areas and massive influx of human populations we are optimistic about the future of Liberty Lake.

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THE CONTINUING DILUTION OF MOSES LAKE, WASHINGTON

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ABSTRACT

The quality of Moses Lake during 1977-79 was markedly improved over that in 1969-70 by the addition of low-nutrient Columbia River dilution water. Total N, total P and chlorophyll *a* improved by 50 percent or greater while Secchi transparency and the fractional composition of blue-green algae in the phytoplankton improved about 40 percent. This resulted from average spring-summer water exchange rates of 10 percent per day in Parker Horn (8 percent of volume) and 0.8 percent per day throughout the whole lake. Although the response of phytoplankton is rather complex, dilution water primarily reduces biomass and chlorophyll *a* by diluting total N in the inflow and causing some instability in the water column through increased circulation, which discourages the accumulation of large blue-green blooms. Other factors such as iron limitation of N fixation in rates and reduction of allelopathic substances may also contribute, but are less discernable in the data. Future control efforts will be oriented toward better distribution of about the same amount of dilution water added throughout the lake in the spring-summer of 1977-79.

INTRODUCTION

Large quantitites of low nutrient Columbia River water have been diverted through existing facilities into Moses Lake, a large (2,853 ha) rather shallow ($\bar{z}$ = 5.6 m) eutrophic lake in eastern Washington. The dilution water has been added to Parker Horn from the Eastlow Canal via Rocky Coulee Wasteway and Crab Creek (Figure 1) at varying rates on seven occasions during the spring-summer of 1977-1979 (Table 1). These inputs resulted in hypothetical lake water exchange or renewal rates of about 10 percent per day for Parker Horn and about 0.8 percent per day for the whole lake, which is about 10 times greater than normal.

Substantial improvement in Moses Lake quality was expected. It was hoped that the total phosphorus content would decrease about 50 $\mu g\ l^{-1}$ and as a result control chlorophyll *a* to an average of about 20 $\mu g\ l^{-1}$. Although it was known that soluble nitrogen was normally depleted and was therefore the apparent growth rate limiting nutrient during summer, total P was expected to better determine biomass because of the prominance of N-fixing blue-green algae. Algal cell loss through simple washout was not expected to appreciably affect biomass.

The results from 1977-1979 that are briefly described here show that total N rather than P probably accounted for much of the 53 percent decrease in chlorophyll *a*. However, the addition of dilution water may also physically deter blue-green algal blooms by *decreasing water column stability*.

Although dilution water was more completely distributed into the Main Arm of the lake than expected, two additional phases of the restoration

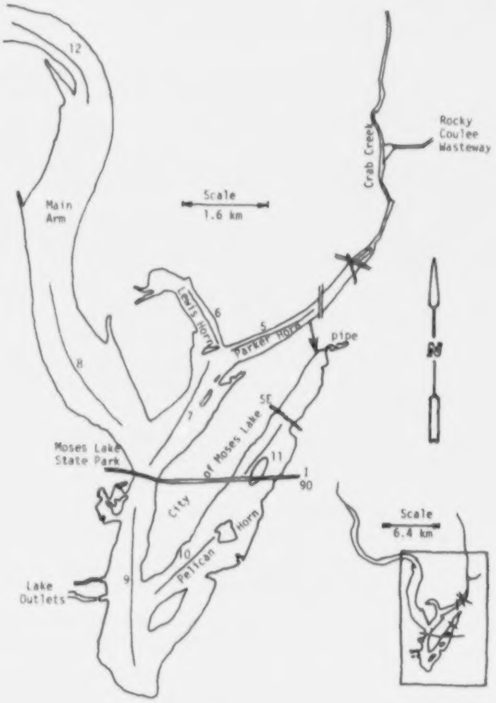

Figure 1. — Moses Lake, Washington: Showing sampling transect locations and treated sewage affluent (SE) and pipe connecting Parker and Pelican Horns proposed in Phase II.

project are planned by Brown and Caldwell Engineering and involve delivering water to Pelican Horn and the Upper Main Arm (Figure 1). Essentially the long-term program for the lake includes a continuous total input of 20 m^3 sec^{-1} from April 1 to June 15, and from 20 to a minimum of 7 m^3 sec^{-1} from June 15 to August 15. The principal change for the long term is to add about the same amount of water during the spring-summer period, but to distribute it more evenly. Total dilution water input has been 203, 111, and 250 × 10^6 m^3 during 1977, 1978, and 1979, respectively. The continuous flows described here amount to an April 1 to August 15 total of either 264 or 178 × 10^6 m^3. The continuous addition, but at a lower rate, is expected to provide at least as good a quality as did the 1977-79 patterns of dilution. The diversion facility from Parker to Pelican Horn should be completed by summer 1981.

Table 1. — Dilution water inflow rates to Parker Horn, Moses Lake via Crab Creek showing hypothetical water exchange rates for Parker Horn (8 percent volume) and the whole lake (100 percent volume) during April through September of the 3 years.

Year	Dilution period	Mean flows in m^3 sec^{-1}		April-September exchange rate in $days^{-1}$	
		dil. water	Crab Creek	Parker Horn	whole lake
1977	3/20-5/07	33.6	0.4	0.11	0.009
	5/22-6/04	10.5	1.3		
	8/14-9/18	17.3	2.5		
	(96 days)				
1978	4/20-6/18 (60 days)	21.7	1.7	0.07	0.006
1979	4/03-6/04	25.1	1.5	0.13	0.010
	7/11-8/28	16.3			
	9/20-10/18	23.2			
	(138 days)				

PROCEDURES

Water was continuously sampled at least weekly along seven horizontal transects in 1977 and eight in 1978 and 1979 at depths of 0.4 meters (Figure 1). The composites were analyzed for total P, soluble reactive P (SRP), nitrate N(NO_3-N), total N, chlorophyll *a*, and specific conductance. Station 12 was added in 1978. At the midpoint of each transect vertical profiles of these variables as well as temperature and Secchi disk depth were determined. The surface values for constituents in the profile were used only occasionally in this analysis. The horizontal transect composites were used as primary data to compare with values from 1969-70, which were determined by the same collection techniques. Procedures for nutrient and chlorophyll *a* analysis followed those of Strickland and Parsons (1972).

Phytoplankton analysis involved counting cells or in the case of blue-greens, unit colonies, in the horizontal transect composites. All counts were converted to volumes based on appropriate cell measurements and geometric configurations. Blue-green count-to-volume conversions were based on procedures by Strathmann (1977). Data from 1970 are based on daily counts made on surface samples taken from a point midway between stations 5 and 7 during four 2-week periods in the summer. Means for those periods were converted to cell volume.

Mean percent of surface light available (I) to phytoplankton was calculated by

$$\bar{I} = \frac{I_0(1 - e^{-KZ})}{KZ}$$

where Z is the depth of mixing estimated from temperature profiles and K is the extinction coefficient estimated from Secchi disk measurements, assuming that the depth of maximum visibility was 10 percent of I_0, the surface intensity. The depth of mixing was either 2 or 4.5 m at station 7 and 1, 2, 6, or 9.2 m at station 9 depending on whether or not the temperature change exceeded 1.0°C.

Percent lake water (% LW) was estimated by assuming that 100 percent LW would be represented by the conductance of Crab Creek water and 0 percent LW by the conductance of Columbia River water. The percent residual LW was thus calculated on each sample date by

$$\% \text{LW} = \frac{\text{LWSC - CRSC}}{\text{CCSC - CRSC}}$$

where LWSC, CRSC, and CCSC are specific conductances for lake water, Columbia River water, and Crab Creek water.

Water column stability was estimated by the change in temperature between surface and bottom at each station and indicated by Δ°C.

RESULTS

Quality Improvement

Marked improvement occurred in nutrient content, phytoplankton biomass composition, and transparency. Total P decreased by 45 percent from a volume weighted mean of 156 μg l^{-1} in 1969-70 to 86 μg l^{-1} in 1977-79, chlorophyll *a* by 53 percent from 45-21μg l^{-1} and Secchi disk transparency increased from 0.9 to 1.5 m. These changes are based on values from all stations except 12 and represent 58 percent of the lake volume for the May to September period. The average rate of water replacement was calculated as 1.4 percent per day for that volume. Degree of improvement was even about 10 percent greater at station 7 in lower Parker Horn (replacement 10 percent per day) in terms of nutrient content and more so in terms of chlorophyll *a*. Of course, chlorophyll *a* in the lower lake was less than in Parker Horn even in the predilution years in spite of little change in nutrient content between the two areas.

Although average nutrient content and chlorophyll *a* were reduced by about 50 percent, large blooms of blue-greens nevertheless occurred in 1977 and 1979 and were characterized by chlorophyll *a* content exceeding 100 μg l^{-1}. The blooms were considerably delayed by dilution, however, compared to the non-dilution years of 1969-70. Once the input of dilution water stopped, phytoplankton biomass began to increase after 3 to 4 weeks. Blooms were not pronounced in 1978 and were delayed much longer — about 2 months.

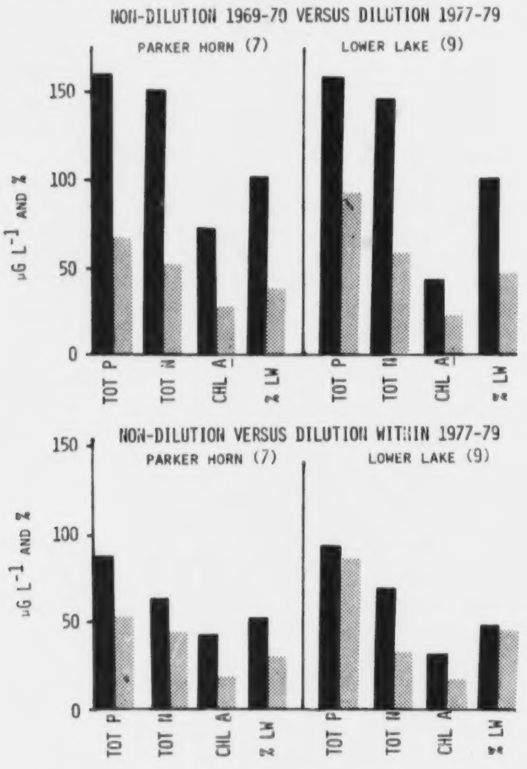

Figure 2. — Mean values for Total P, Total N ( ), chlorohyll *a*, and % lake water (LW) in Parker Horn and the lower lake during dilution (stippled bar) and non-dilution (solid bar) periods between years (1977-79) versus 1969-70) and within years (1977-79).

The rather temporary effect of dilution water input in batches is illustrated in Figure 2 by comparing dilution and non-dilution period means for nutrients, chlorophyll *a*, and % LW. Lag times of 10 and 20 days were allowed in calculating means for Parker Horn and the lower lake. Note that chlorophyll *a* for the non-dilution period is much greater than that for the dilution period in Parker Horn compared to those in the lower lake. That is, recovery of phytoplankton was greater in Parker Horn. Note also that chlorophyll *a* at the lower lake station was quite different between dilution and non-dilution periods in spite of little differences in % LW.

The phytoplankton is still dominated by blue-green algae, especially *Aphanizomenon*, during June through September as it was in the pre-dilution years. However, the fraction of the volume contributed by blue-greens decreased to 55 percent during 1977-79 compared to 96 percent in 1970. Although the observations in 1970 were at a point midway between stations 7 and 5 they are probably comparable. Little difference in composition was noted between stations 7 and 9 in 1977 and 1978. Thus, less biomass coupled with a greatly reduced fraction of blue-greens resulted not only in clearer water, but also in a lower frequency of scum formation (caused by high concentrations of blue-greens).

Controlling Nutrients

Dilution water appears to cause nitrogen rather than phosphorus to be the principal macronutrient controlling phytoplankton biomass. Figure 3 suggests that decreasing the total N to less than 600 µg l^{-1} by adding dilution water reduces the phytoplankton chlorophyll *a*. The N values during non-dilution periods (solid squares) tend to be higher and associated with higher chlorophyll *a* than those during dilution periods. Phosphorus, on the other hand, appears to be less important as a biomass control. Because N values tend to lie to the left and P to the right on the scale of 10N:1P by weight, a reasonable requirement ratio for algal growth, N would appear to be most limiting to biomass formation.

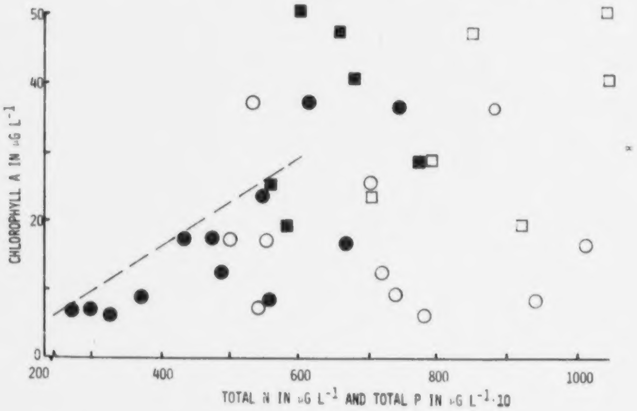

Figure 3. — Chlorophyll *a* related to Total N (closed symbols) and Total P (open symbols) as measured during dilution (circles) and non-dilution (squares) periods at stations 7 and Parker Horn and Lower Lake, with 10- and 20-day lag times, respectively, during 1977-79.

Chlorophyll *a*:cell volume ratios suggest that growth rate was indeed nutrient limited and the soluble N:P ratios indicate also that N was in shortest supply during both pre-dilution and dilution years (Table 2). A few measurements of NH_4-N during maximum growth indicated that the ratios of total soluble N:P were probably as much as four times greater than those listed in Table 2. Chlorophyll *a*:cell volume ratios were near or within the zone of "moderate nutrient deficiency" suggested by Healy (1978). Thus, cell growth was N limited during dilution just as it was prior to dilution, largely as a result of excess P.

Dilution did not necessarily alter the pattern of nutrient limitation but it further restricted the availability of N, the most limiting nutrient, with differences being apparent in the dilution years. For example, total N was substantially less in 1978 and 1979 than in 1977, 535 and 550 versus 600 µg l^{-1} in the lower lake. The ratio of organic N:P (total minus soluble) was also higher in 1977 (12) than in 1978 or

1979 (8 and 9). Phosphorus was initially thought to be the key nutrient to control by adding dilution water to limit biomass. Nitrogen fixation by the most abundant blue-green, *Aphanizomenon,* was considered capable of providing the N needs to match the available P. However, N fixation apparently supplied insufficient N to use the available P; a reduction in total N through dilution therefore effectively lowered average algal abundance.

Table 2. — Mean chlorophyll *a* cell volume ratio, SRP and NO_3 - N during 1977-1979 in lower Moses Lake (station 9). Nutrient means are from May-September data and phytoplankton variables are for June-September.

	chl a: cell volume	NO_3-N $\mu g\ l^{-1}$	PO_4-P $\mu g\ l^{-1}$	$\frac{N}{P}$
1969-70	2.8*	23	48	0.5
1977	4.1	48	56	0.9
1978	2.5	19	24	0.8
1979	1.0	15	26	0.6

* cell volume data from Buckley (1971) at point midway between stations 5 and 7 and chl *a* station 7.

Effect of Washout, Light, and Stratification

Although dilution water input decreased the amount of the most limiting nutrient (N), physical changes are thought to have influenced the persistence of blue-green blooms and in that way also controlled biomass and possibly species composition. One such physical effect is washout, or elimination of biomass from the system at a greater rate than the growth rate can supply new cells. From Figures 4 and 5 it is apparent that on some occasions chlorophyll *a* declined in proportion to % LW after the beginning of a dilution period followed by a recovery and mid-summer bloom. The response of chlorophyll *a* to % LW is most pronounced in Parker Horn where the two variables appear closely related following dilution water inputs in the spring of 1977 and 1979. In the lower lake, however, the response in chlorophyll *a* to a decrease in % LW was much greater than would have been expected from simple dilution or cell washout. In 1977 chlorophyll *a* decreased from 137 to 10 $\mu g\ l^{-1}$ in a period of 1 week while % LW decreased only 25 percent from 52 to 39. In 1979 chlorophyll *a* continued to increase in August while % LW decreased from 46 to 28. The hypothetical water exchange rates in the lower lake during the late summers of 1977 and 1979 were about 5 percent per day. If indeed equally mixed, such rates of exchange are too low to substantially exceed growth rate and account for that magnitude of chlorophyll *a* decrease. However, the surface layer containing the greatest concentration of chlorophyll *a* would tend to be preferentially lost through the lake outlet.

An algal bloom did not begin to develop in 1978 until late August (Figure 6) even though there was only one dilution period following which percent LW recovered as in 1977 and 1979. Furthermore, stratification was as great in June and July of 1978 as in 1977 and 1979.

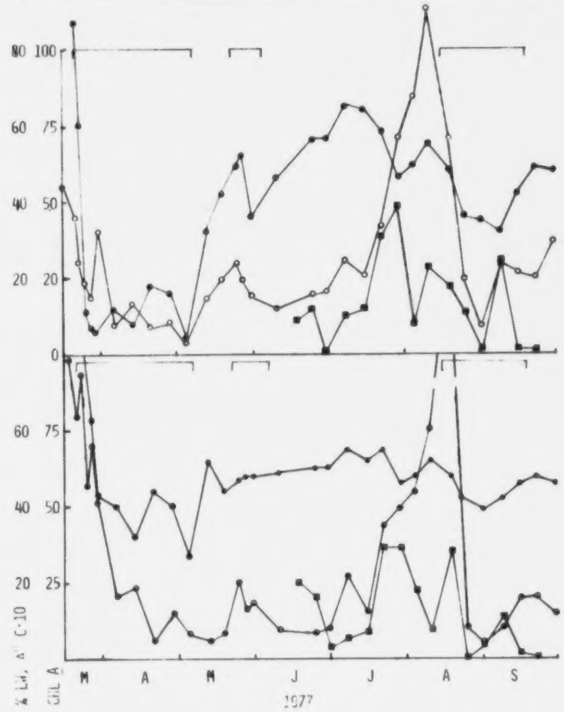

Figure 4. — Chlorophyll *a* (open circles), % LW (solid circles) and Δ°C (squares) in Parker Horn (upper) and Lower Lake (lower) druing 1977. Dilution periods are indicated by brackets.

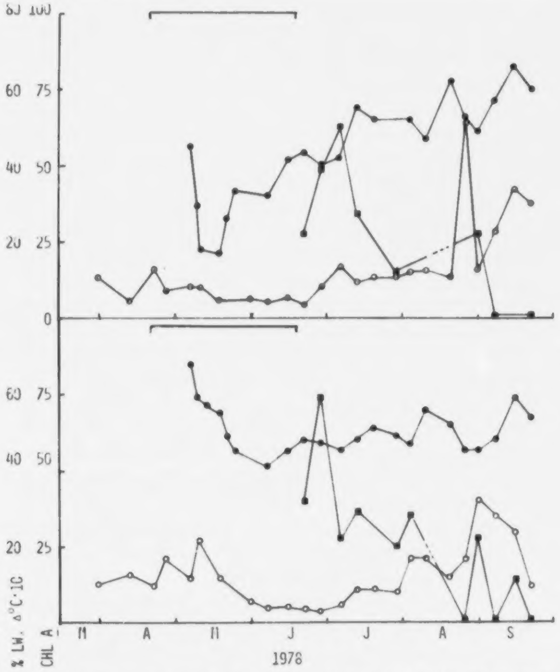

Figure 5. — Chlorophyll a (open circles), % LW (solid circles) and Δ°C (squares) in Parker Horn (upper) and Lower Lake (lower) during 1979. Dilution periods are indicated by brackets.

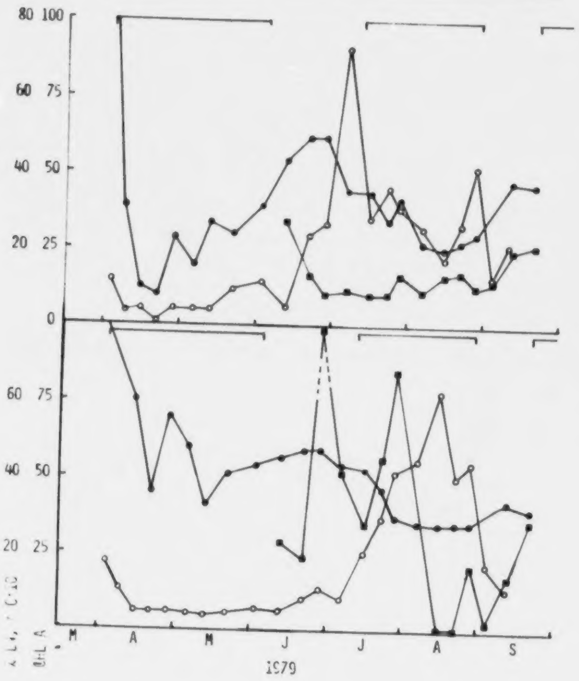

Figure 6. — Chlorophyll *a* (open circles), % LW (solid circles) and Δ^0C (squares) in Parker Horn (upper) and Lower Lake (lower) during 1978. Dilution periods are indicated by brackets.

Physical washout no doubt slowed biomass buildup in Parker Horn during the first dilution inputs in April when exchange rates were 20 to 25 percent per day. However, blue-green algae were shown to grow at a maximum rate of 50 percent per day in 0.5 m deep plastic bags *in situ* (Buckley, 1971). Deliberate exchange of water in the bags at 10 percent per day demonstrated that blue-green increase was minimally affected by physical dilution (washout). This is shown in Figure 6 where the rate of increase in chlorophyll *a* was similar (about 40 percent per day in 100% LW) whether or not exchanged at 10 percent per day.

Light intensity apparently did not greatly influence bloom formation or dieoff during 1977 or 1979. Table 3 shows that I, the average %I (surface intensity) available in the mixed layer, was not different 1 month prior to compared with 1 month after the maximum bloom biomass. Further, the June and July means in Parker Horn and the lower lake were 22, 45, and 37 percent for 1977, 1978, and 1979, respectively. Thus, the failure of a large bloom to develop in 1978 was probably not due to low light levels resulting either from exceptionally well mixed conditions or high extinction coefficients.

The factor that appears most related to bloom development and its subsequent crash is water column stability. Decreases in stability following dilution water input may have been caused largely by the greater mixing produced by the increased rate of water exchange. However, wind influences vertical mixing in Moses Lake and it is rather difficult to separate effects of wind and water exchange rate. Periods of increased stability are normal in Moses Lake, which allows the surface temperature to increase causing even firmer stratification. The buoyant blue-greens tend to be favored by a stable water column and gradually accumulate in the surface layer at concentrations in excess of 100 μg l^{-1} chlorophyll *a*.

Figure 4 indicates that the blooms in both Parker Horn and the lower lake developed under rather stable conditions, but stability tended to break up following the third dilution input and the bloom crash nearly coincided with the decreased stability possibly causing the surface accumulation of chlorophyll *a* to disperse.

The picture was not quite so clear in 1979 (Figure 5). The bloom crash in the lower lake did not coincide with decreased stratification as in 1977, but the 1979 bloom was also not as large as that in 1977. Chlorophyll *a* in Parker Horn in 1979 remained rather high during the dilution period and the water column remained moderately stable as well. Nevertheless, average water column stability, indicated by Δ^0C, was substantially different preceding and following bloom maxima during both 1977 and 1979 (Table 3).

Stability and % LW remained apparently favorable for bloom development during June and July in both Parker Horn and the lower lake in 1978, but for other reasons, including lower total N content, a substantial bloom did not occur.

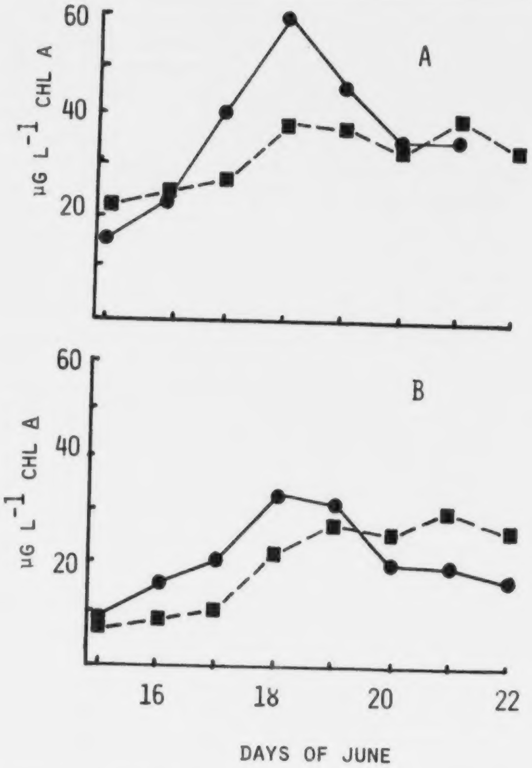

Figure 7. — Response of plankton algae in 100 percent (circles) and 50 percent lake water/50 percent dilution water (squares) without (A) and with (B) a 10 percent per day renewal rate.

Table 3. — Average (± SE) water column stability, percent of surface light (I) and surface (0.5 m) chlorophyll *a* 1 month preceding compared to 1 month following blue-green algal blooms at stations 7 and 9 during 1977 and 1979.

	Chl **a**: $\mu g\ l^{-1}$	Stability Δ^0C	$\bar{I}$, %
pre maximum	58 ± 8	3.0 ± 0.45	24 ± 3.3
post maximum	28 ± 4	1.6 ± 0.45	21 ± 2.8

DISCUSSION

The addition of low nutrient Columbia River water to Moses Lake has markedly improved its quality. Improvements have either equaled or exceeded 50 percent for most variables. This has occurred with rates of water exchange averaging only 1.4 percent per day for nearly two-thirds of the lake. Improvements were slightly greater in the lake section where dilution water entered (Parker Horn) and exchange rates averaged 10 percent per day with maximums of 20 to 25 percent per day. Dilution had a greater relative effect in Parker Horn probably because of the large fraction of soluble nutrients that normally enter with irrigation return flows in Crab Creek.

The exact cause(s) for the overall improvement is not entirely clear. Although N and P totals decreased, soluble fractions throughout the lake were not greatly less and in some cases averaged even more than in predilution years. As in pre-dilution years, N remained the most limiting nutrient for growth rate. The data suggest that total N set the limit on average chlorophyll *a* and probably biomass as well. N fixation by the abundant N fixing blue-green *Aphanizomenon* was apparently insufficient to utilize the available P.

At the rates of water exchange employed, phytoplankton cell loss through washout probably reduced biomass significantly only in Parker Horn where exchange rates during peak spring inflow were at times 20 to 25 percent per day. The decreased biomass in other parts of the lake, where exchange rates during peak inflows were less than 10 percent per day and on the average 1 to 2 percent per day, was probably not caused by washout. On the other hand, increased instability in the water column indicated by small temperature differences with depth (Δ^0C), may have been brought about by increased rates of water exchange. This was suggested by significantly lower Δ^0C, for the 1-month periods following blooms compared to values during the month preceding blooms. However, weekly observations may not realistically reflect the nature and cause of stability, which contributes to or detracts from bloom formation. Although the degree of stability appears related to bloom formation and demise and may have been markedly influenced by dilution water input, much of the instability may also have been caused by wind.

Ahlgren (1979) has suggested that the dominance of blue-green algae in eutrophic lakes may be caused to a large extent by their tolerance of low light and stable conditions. Given sufficient nutrient availability, diatom and green algae biomass could increase until light extinction limited their growth. Blue-greens would then be favored because of their lower light requirements in addition to their buoyancy allowing them to rise in the water column. Under stable conditions, their advantage would be even greater because diatoms and green algae would sink. If rates of water exchange on the order of 5 to 10 percent per day prevent stability in the water column and in so doing discourage blue-green algae, possibly that is an additional benefit of dilution water, even if the added water is not low in nutrients. Eutrophic, productive lakes, in which blue-green algae are not dominant, are not uncommon.

Palmont (1980) suggested yet another cause of biomass control in Moses Lake. Soluble iron was found at concentrations on the order of 1 μgl^{-1} at station 9 in August 1978. Such low iron values are of particular interest when compared with results of bioassays in Clear Lake, Calif., a similar hypereutrophic lake in an arid region. Wurtsbaugh, et al. (1978) found adding ferrous iron to lake water containing 2 to 30 μgl^{-1} dissolved iron greatly increased N-fixation rates and chlorophyll *a*. Dilution water in Moses Lake may have diluted iron, reducing N-fixation. This would have allowed total N flowing into the lake to act more as a control on biomass and chlorophyll *a*. The lower concentration of total N in Columbia River water in 1978 than 1977 (190 versus 360 μgl^{-1}) may have accounted in part for the lower chlorophyll *a* in 1978 than 1977 (16 versus 26 μgl^{-1}). Aside from iron limiting N fixation, the relatively slow rate of N fixation (5 percent per day: Horne and Goldman, 1972) in a system with rather high water exchange rates may have also contributed to the inability of fixation to supply enough N to utilize the available P.

Another potential cause for the dilution water effect in Moses Lake is that of reducing allelopathic substances excreted by blue-green algae. The dilution of such substances, which have been shown to suppress the growth of non-blue-greens (Keating, 1978), may have contributed to blue-greens becoming less abundant in favor of more diatoms and green algae (Welch and Patmont, in press). The difficulty with this hypothesis is that blue-greens represented as sizable a fraction of the biomass in Parker Horn (station 7), where % LW averaged less, as in the lower lake (station 9) where % LW was greater. Although there may be other explanations to the similar fractional contribution of blue-greens in the two lake areas, there is nevertheless no discernible direct relationship between % LW and blue-green fractional composition.

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MANAGING AQUATIC PLANTS WITH FIBERGLAS SCREENS

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ABSTRACT

Vinyl coated Fiberglas mesh screening materials have been used to control nuisance growth of Eurasian watermilfoil (*Myriophyllum spicatum* L.) in selected areas of Lake Washington over the past two growing seasons (1978, 1979). The screens were immediately effective in providing a plant-free water column and greatly retarded regrowth after removal. Two months' coverage in early spring reduced plant biomass by approximately 80 percent throughout the summer. Areas of potential use, methods of application, ecosystem impacts, and economics are discussed.

INTRODUCTION

Lake restoration is often viewed in the context of its relationship to the process of lake eutrophication. Many measures which have been used to mitigate adverse impacts associated with nutrient enrichment of lakes have attempted to alter nutrient loading characteristics. The premise is that longer term benefits should derive by addressing the causes of an observed impact rather than the consequences of that impact. While the rationale is clear, it may not, in all cases, constitute the most practical approach. This would seem particularly true in regard to higher aquatic plants. Perceived nuisance conditions associated with the growth of higher aquatic plants are often related to the introduction and proliferation of exotic plant species and may be localized within discrete areas of a lake. The identification and treatment of distinct causes of such growth may be difficult and perhaps impossible once the plant has become established. Whole lake manipulation to control a particularly troublesome plant may not be necessarily warranted and could have ramifications for the whole system, beyond the initial intent of the manipulator.

A case in point would be the relatively recent infestation of Eurasian watermilfoil (*Myriophyllum spicatum* L.) within selected bays and nearshore areas of Lake Washington. This lake, located in metropolitan Seattle, Wash., represents a classic example of successfully using sewage diversion as a restoration methodology (Edmondson, 1978). Apparently, the presence of milfoil may not be related to nutrient conditions within Lake Washington. While further nutrient input reduction schemes have been suggested, they are not considered to be of practical significance in preventing or controlling the growth and spread of milfoil. Current restoration efforts within Lake Washington are now concentrated on the cosmetic approach of removing the localized nuisance.

A variety of management techniques has been applied to control excessive aquatic plant growth and restore beneficial use within impacted waters. Chemical control techniques using herbicides such as endothall, diquat, and various formulations of 2,4 - dichlorophenoxyacetic acid have largely dominated the aquatic plant management field. As more attention has been directed toward the potential detrimental impacts associated with using chemical control techniques, interest has increased in developing nonchemical alternatives. Mechanical techniques such as harvesting have provided an attractive alternative in those areas where the use of chemicals is limited by either label restrictions or the philosophical attitudes of user groups. Harvesting, while offering potential benefits beyond the simple removal of nuisance plant growth (Carpenter and Adams, 1977, 1978) is limited by water depth, site accessibility, and the requirement for multiple treatments during a single growing season. In some instances, harvesting may also aggravate a situation in that the process may generate an increased number of viable plant fragments which may spread the particular target plant (Kimble, 1980). Harvesting can also lead to the accumulation of a considerable mass of aquatic plant tissue, causing a disposal or reuse problem.

Using bottom barriers as a mechanical means of aquatic plant control has derived largely from work with black polyethylene sheeting materials (Born, et al. 1973; Nichols, 1974). Additional studies have been conducted with a variety of bottom-covering materials and sand/gravel blankets (Nichols, 1974; Armour, et al. 1979; Cooke and Gorman, 1980; Engle, pers. comm.). The results of these applications have varied. In general, where successfully applied, bottom covering would appear to be an effective technique for reducing nuisance conditions associated with the presence of dense aquatic plant growth, at least in the short term.

Problems related to the use of bottom coverings derive from the nature of the sheeting materials used and the modes of application. Most materials described in the literature have relatively low specific gravities which make them bouyant. This bouyancy hinders the application process and renders the material susceptible to lifting by wave action once in place. Even when securely anchored, sheeting materials such as polyethylene tend to be easily torn and dislodged (Armour, et al. 1979). Additionally, unless perforated, most sheeting materials tend to trap gases produced as a result of benthic decomposition which also leads to lifting of the materials from the bottom.

Sheeting materials, whether used by themselves or in conjunction with sand/gravel blankets are usually installed permanently. The accumulation of sediments and detritus on the surface of bottom coverings can be rapid. As this accumulated material constitutes a substrate for continued aquatic plant growth, the effective period of treatment can be greatly reduced. The impacts of these coverings upon benthic invertebrate communities are largely undefined.

The use of Fiberglas screens as a bottom covering was first reported by Mayer (1978). Mayer's work in Chautauqua Lake, N.Y., indicated that many of the problems associated with bottom covering materials were circumvented with the screens. The screen, a negatively bouyant permeable barrier, was highly effective and could be temporarily placed. Larger scale applications and more detailed studies in regard to efficacy, timing and duration of placement, and ecological impacts were conducted in Lake Washington (Perkins, et al. 1979, 1980; Perkins, 1980; Boston, 1980).

LAKE WASHINGTON STUDIES

The screening material used in Lake Washington was the same as that described by Mayer (1978): A polyvinylchloride-coated Fiberglas mesh having 64 apertures per cm², each aperture measuring 1 mm², and a specific gravity of 2.54 (known commercially as Aquascreen). The screens were built to 9 × 24 meters and equipped with grommets at approximately 2 meter intervals along the edges. Concrete reinforcing bar stakes were placed through the grommet holes to secure the panels to the lake bottom.

Treatment plots were delineated within a plant infested embayment at the outlet of Lake Washington (Union Bay). Eight panels were placed in July 1978, the treatment variables being duration of placement (over winter, 1, 2, and 3 months of coverage) and water depth (0 to 2 and 2 to 3 meters). The 1979 work involved the application of three panels in April and three panels in June. Duration of coverage for both the April and June applications were 1, 2, and 3 months for the individual panels. One further application in 1979 involved recovering one-half of two of the 1978 treatment plots. Scuba divers placed all installations.

The effectiveness of Aquascreen was evaluated by following variations in mean dry weight biomass in both treatment and control plots. Estimates of mean dry weight biomass were obtained by randomly selecting five samples from each treatment and control plot at monthly intervals following screen removal. Samples were taken by scuba diver using a cylindrical sampler having a cross-sectional area of 0.25 m². Plant samples were washed free of debris, sorted by species, and oven dried at 60°C for 48 hours prior to weighing.

The results of both the 1978 and 1979 samplings indicated that the screens were highly effective for removing nuisance conditions associated with aquatic plant growth, maintaining a plant-free water column for the duration of placement, and significantly reducing regrowth after panel removal (Perkins, 1980; Boston, 1980). Two to three months of coverage resulted in biomass reductions ranging from 78 percent to virtual elimination of all aquatic plants. One month of coverage provided plant control only for the period of time during which the screen was in place. The results from both years of screening are summarized in Figure 1. Placement of panels in the early spring was more advantageous in that the installation was facilitated by the less dense plant mass occurring at that time and a longer term and more effective biomass reduction was obtained. The results would indicate that screens placed for 2 or 3 months during the period April to June could be removed and

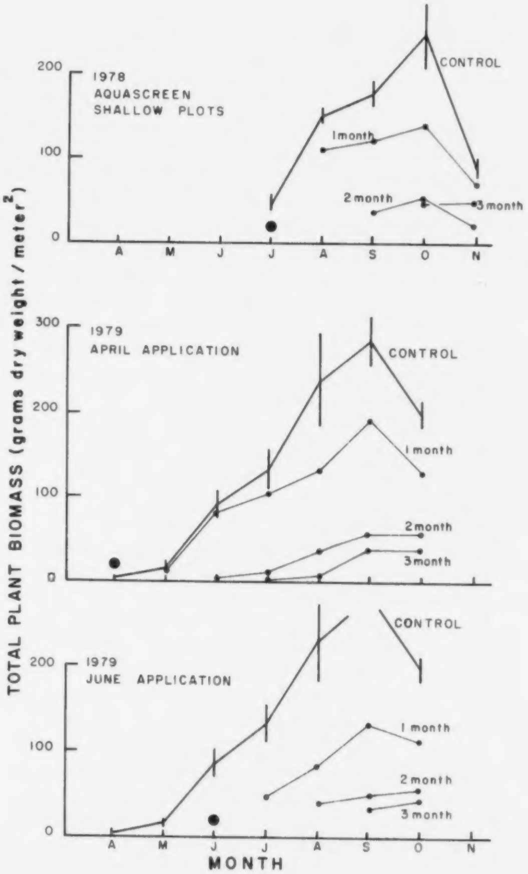

Figure 1. — Variation in dry weight plant biomass within control and screened plots over the 1978 and 1979 growing seasons. Vertical lines represent a 2 SE deviation about the mean for n = 5 replicates.

transferred to another plant infested area, hence doubling the effective area of treatment with a single piece of material.

The release of inorganic nutrients and oxygen-demanding organics as a result of plant decomposition beneath the screens was a potentially significant impact associated with their use. A variety of techniques including field observation and monitoring, field enclosures, and laboratory growth tanks was used to evaluate these factors.

A monthly monitoring of water quality characteristics (dissolved oxygen, pH, alkalinity, dissolved organic carbon, total phosphorus, molybdate reactive phosphorus, total nitrogen, ammonia, nitrate plus nitrite, various cations, and chlorophyll *a*) both within and outside of the screen treatment area failed to indicate any impact of screening upon the water column. The method of assessment, however, was undoubtedly insensitive to such changes if they occurred, as the area was open and subject to fairly rapid water exchange (Perkins, et al. 1980).

To circumvent dilution effects associated with water exchange within the treatment area, we employed 1 square meter polyethylene field enclosures during the 1979 growing season. The enclosures were established in August and monitored at weekly intervals through October.

One and two weeks after the experiment began, dissolved oxygen levels in the screened enclosures were significantly reduced relative to control enclosures. The mean values (±1 S.E. for $n = 3$ replicates) for the screened enclosures were 3.1 ±0.65 and 2.1 ± 1.70 mg $O_2 \cdot \text{liter}^{-1}$ for the first and second weeks, respectively. In two of the enclosures, screen placement was such that plant materials were completely compressed into the sediment; complete compression was not achieved in the third enclosure. In those two enclosures where plant compression was complete, dissolved oxygen rapidly decreased to anoxic conditions (0.0 and 0.9 mg O_2 liter^{-1} after 2 weeks) while the third enclosure did not differ significantly from the controls.

Concomitant with plant decomposition, one might also anticipate increases in the concentration of dissolved organic carbon (DOC) and soluble inorganic nutrients. With the exception of molybdate reactive phosphorus (MRP), such increases were not observed. Concentrations of DOC and soluble inorganic nitrogen in the screened enclosures were not significantly different than the controls. The average concentration of MRP increased by approximately 60 percent relative to the control within 2 weeks. The increase was caused by high concentrations in the two enclosures which went anoxic; the third again did not differ significantly from the controls.

The results of the enclosure studies over the first 2 weeks suggest that when sediment contact is achieved in the installation, plant decomposition is rapid and localized at the sediment-water interface. Presumably, both organic and inorganic materials become entrained within the sediments, creating an increased sediment oxygen demand. Release of inorganic phosphorus to the water column would appear to depend upon whether or not anoxic conditions develop but may also involve the phosphorus retention capacity of the sediments (Boston, 1980).

After the second week, conditions within the screened enclosures rapidly improved. The results of the third week's sampling indicated no significant differences in the various parameters measured. Algal biomass (chlorophyll *a*) increased significantly by the fifth week of sampling and remained at levels higher than the control for the duration of the experiment. These increases occurred in late September and may have been related to increased light availability because the plant canopy was removed.

An assessment of impacts upon the benthic invertebrate community was limited by resources and time. As a result, our efforts were limited to an evaluation of mean densities and composition within treatment plots covered for a period of 3 months only. Samples for identification and enumeration were taken by coring techniques and standard Ekman grab. The results indicated that 3 months of cover with screen had no significant influence upon either mean density (numbers per square meter) or composition of the invertebrate community (Perkins, 1980).

CONCLUSION

Overly dense growth of aquatic plants in lakes creates problems not only for recreational users and private waterfront owners but may also have ramifications for whole ecosystem functioning. This may range from increased rates of sediment accumulation to adverse effects upon the food chain. That aquatic plant management is within the realm of lake restoration seems evident and it is also clear that, whatever technique is applied, it should maintain plant communities intact. This would imply using techniques which could be limited to well-defined areas of need rather than indiscriminant broad scale treatment.

Fiberglas screens would appear to have considerable merit in terms of reducing the nuisance characteristics of aquatic plant growth in a manner commensurate with maintaining the ecological integrity of the system being treated. The screen, if properly applied, can be among the most effective of aquatic plant management techniques. Control can be placed precisely where it is desired, there are no plant disposal problems to contend with, and there need be no adverse impacts associated with plant decomposition. As a nonselective method of control, screen should be used only where complete elimination of plants is the desired end. From this standpoint alone, a limited use strategy would seem most appropriate.

Further, the initial high costs (in excess of $12,000/acre installed) would also argue in favor of the judicious use of screen. By limited use strategy, we are referring to localized nearshore areas where complete control is the desired end and the area of treatment need not be acres. For individual property owners requiring plant control around docks for boat moorage, swimming and diving, or for use on beach areas, the screen seems ideally suited. Season-long maintenance of a 1,000 square foot area with screen could be more effective and accomplished at comparable expense to other management techniques. We have estimated the annual costs for 1,000 square feet of treatment, when allocated over a 5-year life expectancy for the material, to range from $110 to $140.

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RELATIONSHIPS BETWEEN AGRICULTURAL PRACTICES AND RECEIVING WATER QUALITY

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ABSTRACT

Results from several studies will be summarized to more clearly define water quality in typical agricultural areas in relationship to forested or background areas and relationships between agricultural practices and water quality. Studies to be reviewed include a 3-year EPA project analyzing samples from the forested and agricultural piedmont plus well and poorly-drained coastal plain in the Chowan River basin to determine the nature of rural runoff on an areawide basis. Subsequently, a more detailed evaluation of the relationships between agricultural practices and water quality was conducted at four sites in the Chowan River basin, two of which had been in the previous EPA study. The major goal of this continuing study is to determine water quality changes resulting from the best management practices to control agricultural nonpoint sources. As part of the statewide 208 agricultural planning process, three small agricultural watersheds in the coastal plain and piedmont were evaluated to determine relationships between management practices and water quality. Water quality differences between regions and individual watersheds within regions have been analyzed in light of agricultural practices determined from detailed producer surveys within the study watersheds. The study showed that the only significant difference in agricultural activities for watersheds with increased concentrations of nitrogen and phosphorus was greater animal production. Major goals of continuing studies are to document relationships between agricultural practices and receiving water quality and subsequent quality changes.

INTRODUCTION

Assessment and regulation of sources impacting water quality over an entire river basin are complex problems. There are so many rural nonpoint source inputs throughout a drainage basin that complete spatial and temporal evaluation of associated water quality impacts is impractical. Additionally, a basic principle commonly overlooked in areawide water quality assessment is that stream flow from undisturbed lands provides nutrients essential for productive aquatic ecosystems. Therefore, it seems necessary to establish areawide natural or background water quality levels as a basis for assessing the impact of increased inputs from human activities. The tremendous complexity of direct cause and effect relationships between agricultural practices and water quality on an areawide basis make identification of the nature and extent of nonpoint sources very difficult. These dilemmas in no way lessen the importance of water quality planning and evaluation activities to date, but merely emphasize the need to build upon these initial activities to obtain a sound data base for evaluating and directing nonpoint source or rural watershed pollution control programs.

SAMPLING

Historically, judgmental sampling has dominated water quality investigations and evaluations of relationships between agricultural practices and receiving water quality. With judgmental sampling, sites and times are professionally selected as typical of the activities and conditions being studied. This process is characterized by subjective judgment and in this fact lies both the strength and the weakness of this method. The advantages of judgment sampling are that the investigator can select sites and times according to his experience as those best for overall program needs, specific technical requirements, and best use of sampling resources. Disadvantages include the introduction of personal bias in the selection process, the difficulty of determining a sampling error, the fact that the selection process is not reproducible by another investigator because of the personal element, and the lack of a defined universe from which to extrapolate results.

Probability or random sampling has not been used much in studying water quality. Here, after rigorously defining the project scope and specific sampling details, the study sites and sampling times are selected from the total universe being considered. Often the total project may be subdivided into smaller, nonoverlapping but all-inclusive parts which can be sampled independently.

Probability sampling has been widely applied to many scientific and social problems. Its advantages are that unbiased estimates may be derived by reproducible methods; inference may be made from the sampling results to the defined universe; a statistical sampling error can be estimated; and with certain assumptions about statistical distribution, confidence limits may be set about the estimates. The ability to estimate sampling error provides a rational basis for considering the optimal allocation of sampling effort.

Disadvantages of probability sampling are that without effective stratification, too much of the field effort may be spent at the sites and times, which do not result in maximum efficiency for achieving study goals. Some randomly selected sites may seem inappropriate or nonrepresentative but yet every site is unique in some way. Additionally, some randomly selected sites cannot be sampled but because of this should not have been included in the sampling frame.

Either automated or grab methods may be used to obtain water samples and measure flow. Judgmental or probability sampling may be used to direct these sampling strategies. An automated installation is costly but it can provide a continuous record of stream flow and a programmed series of water samples. Grab sampling costs less per site and is therefore a more flexible method when covering many sites, but it provides information about flow and water quality only for the time of the sampling visit.

STUDY ON SAMPLING TECHNIQUES AND AREAWIDE WATER QUALITY

A 3-year EPA project entitled, "Probability Sampling to Measure Pollution from Rural Land Runoff," (Humenik, et. al. 1980) investigated the feasibility of using probability sampling in describing rural water quality not affected by point sources on an areawide basis. The study also examined the substantive results of this sampling effort for greater insight into relationships between land activities and receiving water quality.

The Chowan River Basin which was selected as the study area is about 209 kilometers long and drains an area of 12,802 km² in southeastern Virginia and northeastern North Carolina (Figure 1). The upper 5,180 km² of the basin lie in the gently rolling hills of Virginia's piedmont plateau. The remaining 7,700 km² lie in the flat coastal plain of Virginia and North Carolina where the soils can be broadly classified as either poorly drained or well-drained sands. Within the piedmont there was no logical reason for stratification by soil type because nearly all the soils are loamy. However, because water quality differences are likely

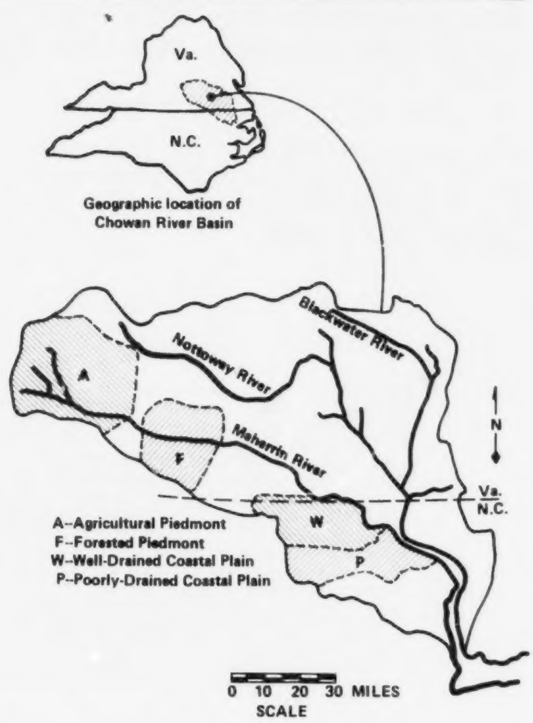

Figure 1. — Study basin location and delineation of sampling areas.

Table 1. — Land use in Chowan river study subbasin.

Subbasin	Drainage area sq mi	Drainage area sq km	Forest %	Crop %	Pasture %	Developed %	Logged %	Ponds %
			Poorly-Drained Coastal Plain					
P-8	4.51	11.68	77.1	17.4	4.2	1.0	0.3	0.0
P-10	3.74	9.69	72.1	25.9	2.0	0.0	0.0	0.0
P-11	4.90	12.69	69.0	23.0	3.0	4.0	1.0	0.0
P-13	38.04	98.52	66.2	26.4	4.7	1.5	1.1	0.0
			Well-Drained Coastal Plain					
W-3	6.27	16.24	44.7	53.5	1.3	0.3	0.2	0.0
W-4	0.20	0.52	52.6	46.2	1.2	0.0	0.0	0.0
W-8	3.31	8.57	56.2	41.9	0.4	1.5	0.0	0.0
W-10	6.37	16.50	48.5	43.3	7.0	0.9	0.3	0.0
			Forested Piedmont					
F-1	5.21	13.49	72.1	15.0	8.1	0.0	4.8	0.0
F-2	6.14	15.90	91.9	3.7	2.8	0.2	1.4	0.0
F-3	14.06	36.42	90.3	7.0	1.0	0.1	1.6	0.0
F-7	6.04	15.64	82.8	10.6	3.0	2.9	0.7	0.0
			Agricultural Piedmont					
A-1	5.57	14.43	63.6	27.0	7.3	0.2	1.7	0.2
A-4	4.28	11.09	55.6	29.9	14.3	0.2	0.0	0.0
A-8	1.75	4.53	38.1	20.6	32.5	1.0	7.6	0.2

to result from different land uses, a forested land was selected to represent background conditions and an agricultural area was selected to measure land-use impact. Thus the four study areas were forested piedmont, agricultural piedmont, poorly-drained coastal plain, and well-drained coastal plain. The areas designated for sample site selection were from a restricted part of the watershed to allow more convenient sample retrieval; they represented about 25 percent of the total watershed area.

The sampling site was defined as a point located at the first railroad or road crossing below the confluence of two first-order streams on a U.S. Geological Survey 1:250,000 map above which there were no sources that would require discharge permits. All such sites (above 90) that met the definition within the study areas were identified and comprised the sampling universe. Then by random selection four sites were chosen from three of the areas and three sites from the fourth area for a total of 15 sampling sites. The drainage area of these sites ranged from 0.5 to 100 km^2 and the general land use ranged from 40 to 90 percent forested (Table 1). Such stream channels were from 1.5 to 15 meters wide and at base flow from 0.15 to 0.60 meters deep.

Two years (November 1974-November 1976) of data were collected at the forested piedmont, poorly-drained coastal plain, and well-drained coastal plain sites, and 18 months (June 1975-November 1976) of data were collected at the agricultural piedmont site. The grab sampling plan involved time stratification to ensure that measurements were obtained at a uniform rate throughout the study. Basically, the stratification was such that each stream was monitored 26 times per year at the rate of two visits (chosen by a restricted random sampling) per 28-day period. During each grab site visit the flow rate was measured by standard USGS procedures and a depth-integrated water sample was obtained manually at the midpoint of the stream.

Automated sampling systems were established at 5 of the 15 statistically selected grab sampling sites. Stream stage was recorded continuously by analog recorders and the automated samplers had the capability of collecting 28 discrete 500-ml water samples. The sampler was activated by stage change with a subsample to be taken at each 76 mm rise or fall in stream stage. The time each subsample was obtained was recorded on a stage strip chart by a relay activated pen so that each sample bottle was assigned the mid-point time between samplings for mass transport analyses.

COMPARISON OF GRAB AND INSTRUMENTED SAMPLING DATA

Many water quality assessment agencies have limited monetary resources so that the grab sample approach is often used to determine regional water quality. The sample frequency varies, but typically the sample design is such that the stream is periodically instead of randomly sampled. The 22-month data base from one poorly drained coastal plain site that had both grab and instrumented sampling was employed to compare the results of such sampling on flow and concentration estimates. For illustrative purposes the value obtained from the instrumented sampler for the 22-month period was also plotted at the 24-sample per year frequency (Figures 2 through 6). As anticipated, the range of grab sampling values generally increased with decreasing sample frequency. This trend is not surprising and can be predicted by sampling theory. Additionally, in some cases the high frequency grab sampling value is very similar to the instrumented value, and in other cases markedly different. Realizing such parameter variation with space, time, and sampling costs as a function of the number of sites and visits to each site is most important in developing a technically sound monitoring scheme within given budget requirements.

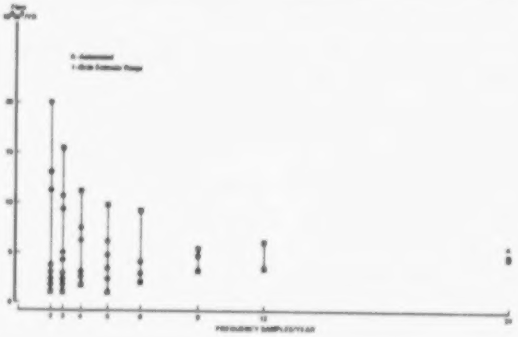

Figure 2. — Range of Mean Flow Velocity as a function of Grab Sample Frequency.

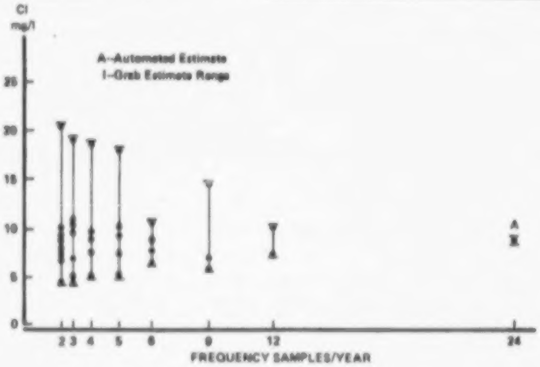

Figure 3. — Range of Mean Chloride Concentration as a function of Grab Sample Frequency.

Data from the five sites which were both instrumented and grab sampled were compared for the total 22-month sampling period. Constituent mean values at all sites indicated that differences existed ($p < 0.10$) between chemical oxygen demand (COD), total organic carbon (TOC), total phosphorus (TP), total Kjeldahl nitrogen (TKN), chloride (Cl^-) concentrations measured by grab and automated samples while there was no evidence of a difference ($p < 0.10$) for nitrate (NO_3^-) concentrations. The grab mean concentrations were less than the automated mean concentrations by

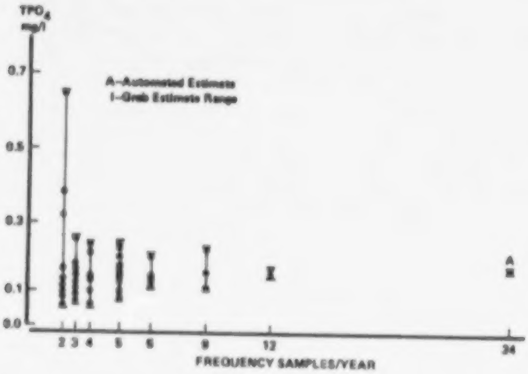

Figure 4. — Range of Volume Average Mean Total Phosphorus Concentration as a function of Grab Sample Frequency.

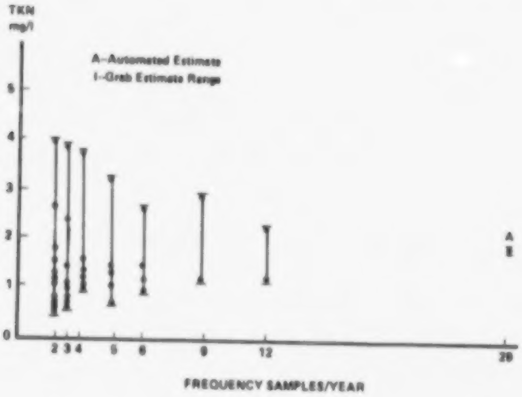

Figure 5. — Range of Mean Volume Average Total Kjeldahl Nitrogen Concentration as a function of Grab Sample Frequency.

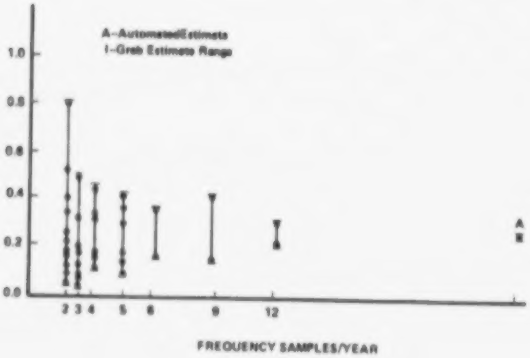

Figure 6. — Range of Mean Volume Average Nitrate Concentration as a function of Grab Sample Frequency.

36, 39, 28, 19, and 8 percent for COD, TOC, TP, TKN, and Cl, respectively.

Relatively large differences were observed between grab and instrumented estimates of annual volume average concentration for a given site. Because the grab to automated annual volume average concentration ratio varied among constituents at a site, no consistent factor related the two estimates. Further, for each water quality constituent the grab to automated concentration ratio varied among the sites; thus a factor relating the two estimates which was independent of site did not exist. Statistical testing indicated ($P < 0.10$) that annual volume average grab concentration estimates of COD, TOC, and TP which were about 50 percent less than automated sampling values were statistically significant. For TKN, NO_3^- and Cl^- grab sampling was less by an average 13 percent, not significant statistically.

The annual grab and automated water yield estimates obtained at different sampling sites resulted in relatively large differences between paired samples. However, statistical testing for paired samples provided no evidence of a statistically significant difference between estimates by the two methods.

Conclusions

Statistical sampling methods can be used to measure the mean areawide contribution of chemical species from rural nonpoint sources as an alternative to the more difficult and often impractical complete monitoring approach. Grab and instrumented sampling are two common methods of assessing stream water quality which can be employed in a statistically designed sampling program. Although differences were found for grab and instrumented sampling estimates of concentration and a flow resultant data, analyses from both sampling methods supported the same general conclusions.

Annual volume average concentration and water yield estimates were obtained at five sites by routine operation of both stage activated, instrumented samplers with stage recorders and simple time stratified grab sampling. Results indicated that about 50 percent lower COD, TOC, and TP estimates were obtained by the grab sampling method. Although rather large differences were observed for the water yield estimates, the data did not indicate any statistically significant difference between the two methods. Due to the confounding factors associated with the two sampling methodologies, it was not possible to completely define reasons for the differences.

Comparing annual water yield estimates from the 15 statistical survey sites to historic values for the study region verified that simple time-stratified grab sampling provided reasonable estimates of areawide annual water yield; however, the precision of the individual site estimates was low.

RELATIONSHIPS BETWEEN LAND USE AND WATER QUALITY

Prediction of present and future rural water quality is often based upon models which relate water quality to macro land-use factors. These models are probably best supported by a national water quality land-use study of 928 sites reported by Omernik (1977). That study found nitrogen and phosphorus concentrations increased with increased agricultural intensity. Water quality land-use relationships for the 15 statistical survey sites were investigated to determine if similar relationships were evident in this watershed. These analyses considered both in-stream conditions and exported water quality as a function of percent forested area. The land-use summary presented in Table 1 for these subbasins showed that the forested area, plus agriculture area, was approximately equal to the total subbasin area at each site. Thus in these subbasins decreased forested area indicated increased human activity which was primarily agriculture. Therefore, major trends which were observed for water quality constituents with respect to percent forested area would also exist (only inversely) with respect to percent agriculture.

Site water quality versus land-use relationships for COD, TP, TKN, and NO_3^--N concentrations during June 1975-November 1976 are presented in Figure 7. For each of the four parameters, the grab sampling mean, a mean plus or minus one standard deviation, maximum, and minimum values for each site are plotted as a function of percent forested area. From a water quality perspective, the COD, TP and TKN vs land-use graphs show no meaningful mean concentration increase as the percent forested area decreased. Indeed, even the range of values for these parameters was relatively constant for all sites. The mean NO_3^--N concentration also does not display a concentration increase with decreased forested area, but the values vary considerbly from site to site, indicating that the sites were probably differentially impacted by the general land-use activities. The major conclusion obtained from examining these graphs was that the mean time average concentration was relatively uniform throughout the watershed and did not increase as agricultural activity increased.

The effects of land use on receiver system water quality was assessed by analyzing flow weighted concentrations of COD, TP, TKN, and NO_3^--N versus percent forested area for the 15 statistical survey sites during June 1975-November 1976. These data are displayed in Figure 8. The flow weighted concentrations do not present any clear relationship between water quality and percent forested area so regression models were not attempted.

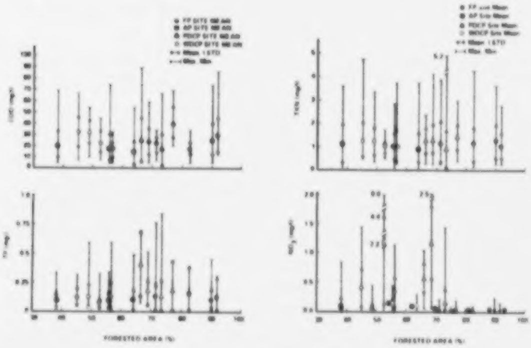

Figure 7. — Site arithmetic data summary versus land use for grab sampling (June 1975 to November 1976).

Nutrient levels in streams usually increase as agricultural intensity increases, but often a wide range of confidence limits exists for regression models developed for large geographic areas. For example, one regression model (Omernik, 1977) relating mean total phosphorus concentration to percent agriculture plus urban area for the eastern United States had broad confidence limits as measured by a ratio for the plus or minus 1 sigma range to predicted mean value of 130 percent. The Chowan concentration versus land-use graphs point out the need for caution when employing model predictions to specific cases. Regression models employing macro land-use factors do not account for varying agricultural cropping patterns and management practices, annual weather conditions, stream border buffer systems, or other factors which can impact agricultural effects on water quality.

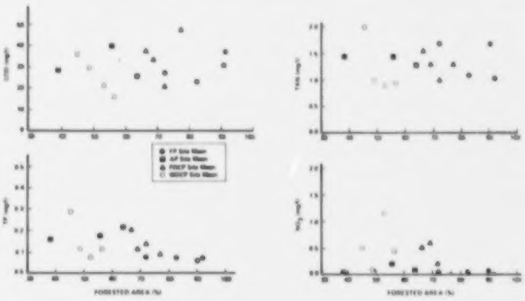

Figure 8. — Site flow weighted concentrations versus land use for grab sampling (June 1975 to November 1976).

Conclusions

While direct water quality land-use relationships for the Chowan data were weak, differences related to geographic area, season, and size were observed. Evaluation of the water quality data obtained during this study led to the following conclusions concerning nature of rural runoff on an areawide basis as developed and summarized in the project report (Humenik, et al. 1980).

1. Neither in-stream (arithmetic average) nor net export (flow weighted) concentration data presented any clear relationships between water quality and macro land-use factors as measured by percent of forested land. This result points out the need for caution when applying model predictions to specific cases. Macro land-use factors do not account for varying agricultural cropping and management practices, annual weather conditions, stream border buffer systems, or other factors which can minimize the impact of agricultural activities on water quality.

2. The comparison of geoclimatic areas demonstrated that the dominant variation was between the piedmont and coastal plain with only minor variations occurring within these two physiographic regions. The

differences between the piedmont and coastal plain were judged to result from naturally occurring physiographic variations in (a) basin characteristics such as vegetation, soil type, and stream hydraulics; and (b) ocean proximity.

3. Average stream water quality for the four sampling areas was relatively uniform. Quality was generally good compared to proposed standards, but elevated TP concentrations even in the forested piedmont area demonstrated the need for basing water quality assessments on measured local conditions, especially background or natural levels. Stream sample concentrations usually displayed large variations with respect to mean values indicating that rural nonpoint sources are highly variable in both space and time.

4. Analyses for seasonal trends indicated that water yield and the associated nutrient yields were greater during the winter and spring seasons than during the summer and fall, reflecting rainfall and evapotranspiration cycles. In analyzing seasonal flow weighted average concentration data, the models generally demonstrated significant relationships but had rather low r^2 values.

5. Measured concentrations did not display any consistent functional relationship to flow (water yield) levels. However, data showed that NO_3-N concentrations were elevated during flow conditions at a small (0.5 km^2 or 0.2 mi^2)site but not at a larger (20 km^2 or 8 mi^2)site but not at larger (20 km^2 or 8 mi^2) site in the well-drained coastal plain with similar land use. The NO_3-N attenuation was judged to be the result of in-stream dynamics.

6. The impact of channelizing coastal plain streams was most pronounced with respect to high NO_3^--N concentrations in the channelized streams as compared to the unchannelized streams which have natural swampy flood plains and channels that increase NO_3-N attenuation by denitrification and biological uptake.

7. Assessment of point and nonpoint source impacts in one small basin verified classic point source concentration spikes with subsequent decline to intermediate levels for all investigated constituents except chloride and nitrate. Therefore, for the studied stream reach, nitrogen and phosphorus inputs which appeared to come from treatment plant effluents are reduced to headwater background levels as long as stream assimilatory capacity is not overwhelmed or natural inputs change background levels.

8. Point-in-time comparisons between headwater and downstream constituent concentrations showed small differences on a water quality basis.

208 CRITICAL AREA STUDIES

Two areas of intensive agricultural production in major North Carolina river basins were selected for critical area studies under the statewide 208 water quality planning and implementation program for agriculture. Within both the coastal plain and piedmont study area three predominantly agricultural watersheds between 13 to 26 sq. km were randomly selected for intensive monitoring (Horney, et al. 1978).

The coastal plain study subbasins in the Neuse River have intensive cropping (predominantly corn, tobacco and soybeans) and considerable swine production. The selected watersheds have slopes ranging from 1 to 6 percent with sandy loam and its associates.

The piedmont study area subbasins have steeper topography and slate soils that increase the significance of erosion and sedimentation. Slopes of the study watersheds generally range from 4 to 10 percent although some as steep as 18 to 20 percent were found. Major crops are corn, sorghum, and soybeans. Some land is in permanent pasture for beef production, but swine and poultry production predominate.

Instrumented sampling stations were installed at the lower boundary of each watershed. Rainfall was measured by recording rain gage and a digital stage recorder measured stream stage from which flows were determined. Water samples were taken during storm events by an automated sampler adapted to sample across the runoff hydrograph (Koehler, et al. 1978). In the coastal plain study area a background station had been established by USGS as representing undisturbed forest lands with an area of 2 sq. km of which 95 percent is forested, 5 percent is in agriculture, and less than 1 percent is paved roads. This site was grab sampled during storm events over the study period.

Data Analysis

Concentration data from the three monitored watersheds in each study area are combined in Table 2. Data from the coastal plain background station are also presented to allow comparison with storm grab sampling data for such a relatively undisturbed area. Data are included from State monitoring stations on the river stems closest to the priority areas in the two study watersheds. These main river stations were grab sampled quarterly and therefore do not represent runoff conditions.

Mean concentrations in the study streams draining agricultural areas were higher than the background and river stations. As expected, the background station had the lowest mean constituent concentrations. Since the watershed sampling program was designed to measure water quality during runoff conditions, these concentrations can be attributed mainly to rainfall runoff transport. Study area runoff concentrations were higher than the major receiver stream average flow concentrations by a factor of 1.4 to 4 times in the coastal plain and 5 to 10 in the piedmont.

Average data for all sites in a region showed mean concentrations for all constituents were higher in the piedmont than the coastal plain study areas (Table 2). An analysis of variance was performed for nitrogen and phosphorus data to determine whether the regional differences in mean concentrations were significant. While statistically high for all N and P forms, only 9 to 23 percent of the variance within the data was due to the regional separation; therefore, there must have been considerable variation between and within individual watersheds over time.

Water quality data for each site collected from April 1978 to June 1979 in the coastal plain study areas and July 1978 to June 1979 in the piedmont study areas are summarized in Table 3. Median, maximum, and minimum concentrations are included as well as the

Table 2. — Mean constituent concentrations (mg/l) in study area, background, and river stations.

Parameter	Coastal Plain - Background (n* = 23)		Coastal Plain - Watersheds (n = 314)		Coastal Plain - River (n = 7)		Piedmont - Watersheds (n = 261)		Piedmont - River (n = 12)	
	Mean mg/l	S.D.*	Mean mg/l	S.D.	Mean mg/l	S.D.	Mean mg/l	S.D.	Mean mg/l	S.D.
Ammonia Nitrogen (NH_3-N)	<0.05	0	0.21	0.62	0.05	0.03	0.64	1.61	0.07	0.06
Total Kjeldahl Nitrogen (TKN)	0.18	0.14	0.90	1.44	0.56	0.08	2.36	3.02	0.31	0.12
Nitrite + Nitrate Nitrogen (NO_2-NO_3-N)	0.13	0.26	0.93	1.34	0.70	0.31	2.27	2.48	0.62	0.29
Total Nitrogen (TN)	0.30	0.28	1.83	1.97	NA*		4.63	4.46	NA	
Total Phosphorus (TP)	<0.05	0.01	0.32	0.52	0.22	0.06	0.51	0.64	0.09	0.10
Total Residue (TR)	52.0	24.0	232.0	706.0	NA		901.0	2005.0	119.0	23.0
Total Nonfilterable Residue (TNR)	7.0	7.0	39.0	69.0	NA		617.0	1397.0	NA	
Chemical Oxygen Demand (COD)	12.0	9.0	46.0	34.0	26.0	6.0	63.0	63.0	9.0	5.0

*S.D. - Standard Deviation
NA - Not Available
n - Number of Observations

Table 3. — General statistics for water quality parameter concentrations (mg/l) in each monitored watershed.

Parameter	CP-1 (74 Observations)					CP-2 (100 Observations)					CP-3 (99 Observations)				
	Median	Mean	S.D.	Min	Max	Median	Mean	S.D.	Min	Max	Median	Mean	S.D.	Min	Max
NH_3-N	<.05	0.05	0.07	<.05	0.46	0.16	0.48	1.04	.05	7.70	0.08	0.09	0.07	<.05	0.37
TKN	0.40	0.46	0.21	0.20	1.70	0.60	1.51	2.36	0.10	19.0	0.60	0.64	0.24	0.30	1.40
NO_{2+3}-N	0.65	0.71	0.38	<.05	1.50	1.50	1.68	2.10	0.13	22.0	0.49	0.61	0.57	<.05	1.90
TN	1.07	1.18	0.50	0.33	2.60	2.50	3.19	2.97	0.54	22.3	1.09	1.24	0.52	0.43	2.40
TP	0.06	0.07	0.06	<.05	0.39	0.26	0.58	0.78	0.05	4.80	0.17	0.24	0.22	0.06	1.40
TR	70.0	449.0	1179.0	40.0	4650.0	93.0	121.0	73.0	41.0	429.0	98.0	243.0	736.0	38.0	5150.0
TNR	12.0	13.0	11.0	0.0	54.0	22.0	47.0	61.0	1.0	369.0	24.0	50.0	92.0	4.0	670.0
COD	38.0	40.0	13.0	22.0	94.0	43.0	58.0	47.0	12.0	280.0	31.0	35.0	20.0	17.0	180.0

Parameter	P-1 (80 Observations)					P-2 (81 Observations)					P-3 (66 Observations)				
	Median	Mean	S.D.	Min	Max	Median	Mean	S.D.	Min	Max	Median	Mean	S.D.	Min	Max
NH_3-N	0.23	0.33	0.38	<.05	1.80	0.56	1.13	2.05	<.05	15.0	0.11	0.16	0.17	<.05	0.85
TKN	1.50	2.65	2.63	0.30	15.0	1.60	3.23	3.84	0.40	18.0	0.90	1.23	0.79	0.20	4.60
NO_{2+3}-N	1.80	3.15	3.11	0.27	18.00	1.30	2.51	2.40	0.20	11.0	0.66	0.91	0.78	<.05	3.80
TN	4.10	5.79	4.61	0.77	21.2	3.52	5.78	5.13	0.98	28.0	1.66	2.04	1.42	0.34	8.40
TP	0.39	0.85	0.82	0.06	3.00	0.34	0.53	0.62	0.05	4.0	0.18	0.26	0.20	<.05	0.97
TR	283.0	1669.0	2346.0	88.0	9620.0	217.0	889.0	2514.0	81.0	21700.0	173.0	416.0	707.0	60.0	4040.0
TNR	128.0	1326.0	2091.0	6.0	9330.0	83.0	536.0	1172.0	5.0	6810.0	86.0	185.0	275.0	4.0	1730.0
COD	47.0	93.0	84.0	14.0	290.0	41.0	59.0	58.0	17.0	380.0	38.0	42.0	20.0	1.0	110.0

standard deviation to indicate data distribution. Similar mean and median values indicate a fairly even distribution about the mean while a mean value considerably higher than the median indicates a few extreme values.

An analysis of variance was performed to determine whether differences among watersheds within each study area were significant. Differences were found to be statistically high for the nitrogen and phosphorus forms with a variance of 35 to 48 percent. This indicates that water quality differences are a function of the individual watershed as well as regional location. Over 50 percent of the variance is not explained by this classification indicating the highly variable nature of small streams as reported from a previous study (Humenik, et al. 1980).

Conclusions

General conclusions resulting from this study as developed in the total project report (Horney, et al. 1979) are:

1. Streams draining predominantly agricultural watersheds have higher nitrogen and phosphorus runoff concentrations than in undisturbed forested watersheds in the same area and long-term averages in receiver river streams.
2. Concentration levels are generally higher in the piedmont than in the coastal plain; however, variation between individual watersheds within regions are as important as regional differences.
3. The watershed in both the piedmont and coastal plain study areas with the highest nitrogen and phosphorus concentrations also has the highest level

of livestock production. Since no direct animal waste point sources were found, many such waste inputs would have been from nonpoint sources.

4. Excessive use of chemical fertilizers was found in some fields in all areas, but crop management practices were found to significantly differ from one watershed to another and thus relationships to water quality differences could not be determined on the basis of cropping patterns.

Analysis of nitrogen forms indicated possible differences in nitrogen transport related to observed livestock production systems and waste management techniques. Better understanding of these sources and transport mechanisms should result from a more complete analysis of production-waste management practices and stream flow data.

SUMMARY

The relatively similar average concentrations from sampling sites receiving rural nonpoint source inputs from different land use and geoclimatic regions and in main rivers draining these sites indicate that more data on background conditions and relative impact of nonpoint sources are needed before wide implementation of best management practices is required, particularly in areas with heterogenous land use. It also seems most important that agencies developing regulatory criteria be responsive to ambient conditions and not entertain standards requiring better than background water quality particularly for relatively undisturbed or pristine areas such as the forested regions evaluated in these studies.

Tools must be developed to assess areawide relationships between agricultural practices and receiving water quality. Efficient use of currently available sampling and modeling techniques can provide more cost-effective assessment of both management practices to control agricultural nonpoint sources and areawide water quality over time and space for existing conditions and different planning strategies. The effects of agricultural practices on the water quality of both a stream reach and an area must be defined to answer the important and difficult questions concerning cost effectiveness and technical feasibility of management practices to control agricultural nonpoint sources and achieve areawide water quality goals.

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SOURCE CONTROL OF ANIMAL WASTES FOR LAKE WATERSHEDS

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ABSTRACT

Controlling wastes from animal production facilities is conceptually well defined. There are many components of an animal waste management system that have been proven with use. The trick to designing a successful system for a specific location is to know what component will work in that climate and under the management regime of the farm. Containing these wastes is basically an engineering problem; the disposal or utilization becomes an agronomic problem, and the resultant runoff or pollution from disposal is an environmental problem. All of these must be addressed in the planning stage and not after the fact. Controlling animal wastes produced from a nonpoint source or pasture production system becomes a more difficult problem. This type of control relates almost totally to pasture management and livestock management with very little engineering involved. Therefore, the solutions are termed best management practices (BMP) and are selected from a set of management practices proposed for a region of the United States.

NATURE OF ENVIRONMENTAL PROBLEM

Livestock and poultry industries are a significant part of the U.S. economy. We currently have approximately 44 million beef cows, 11 million dairy cows, 10 million beef feeders, 50 million swine, 16 million sheep, 400 million layers, 460 million broilers, and 46 million turkeys.

Livestock and poultry industries annually produce about 112 million tons of dry residue to be applied to crop or pasture land. Contained in this residue are about 4.1 million tons of nitrogen (N), 1.1 million tons of phosphorus (P), and 2.4 million tons of potassium (K). U.S. agriculture uses about 9.2 million tons of chemical fertilizer N annually; therefore, it is easy to see the commercial impact of animal wastes if it could all be collected and used to replace some of the chemical fertilizers. Further information can be obtained from a USDA publication entitled "Estimating U.S. Livestock and Poultry Manure and Nutrient Production."

VARIETY IN TYPES OF ANIMAL PRODUCTION UNITS

In the beef cattle industry only one-fourth of the 54 million animals are in confined feedlots. The remainder are on pasture or in some partial confinement system, such as winter feeding of cows. About 95 percent of those confined are in open dirt lots; of the 137,700 feedlots in operation, less than 20 percent are under the restrictions of the effluent guidelines. Most of the feedlots store manure in open piles for application to land during the spring and/or fall. Only about 1 to 2 percent of the nearly 400,000 dairy farm units are subject to discharge permits. About 25 percent of the farms use milking centers and cow yards or pastures; the other 75 percent use some type of confinement. Most of the confinement systems use daily cleaning, and the manure is quite often spread daily regardless of weather conditions, i.e., frozen ground and/or snow cover.

Hog production in the U.S. is about equally divided into open dirt lot production and covered, paved units, with most in units of less than 200 heads. The wastes from many of the new housed production units are liquid in nature and are stored in lagoons or pits for later land application.

Poultry are nearly all produced in housed units, turkeys being the exception with nearly 75 percent of them on range or in open confinement pens. Most of the collectable waste from housed units is in a solid form, with only a few units using liquid manure systems.

IMPACT ON WATER QUALITY

The effect of animal wastes on water quality has been known for many years but has been dramatically demonstrated during the past 20 years, when fish kills were related to the runoff from feedlots entering the streams. However, the relationship between land application of manure and the water quality of a stream receiving runoff from the application site has not been fully established. The effect on chemical water quality can be estimated by using dilution factors or by actual stream quality measurements. Such stream water quality measurements are expensive; few have been made in the past, and only recently have large scale studies been initiated. To actually study the nonpoint source component of the stream quality in a large basin would be very expensive. Moreover, effects on the biological community within receiving waters are unknown for the most part; this subject needs considerable effort in the near future.

Annual pollution loads resulting from land application of animal wastes change dramatically because of the management practice selected. The magnitude of such a change can be illustrated by using the entire State of New York as an example. Assuming that New York has approximately 900,000 dairy cows producing enough manure to supply the nitrogen needs of 450,000 acres of cropland, the following two management practices demonstrate the difference in annual nitrogen loading to the waters of the State. If this waste is applied daily without incorporation into the soil, the total load of nitrogen to the waters of the State would be approximately 3.6 million pounds. When the assumed best management practice of proper application timing and total incorporation of the waste is evaluated, the nitrogen load to the waters would be only 0.8 million pounds annually. This would be a reduction of 2.6 million pounds of nitrogen annually, which could have a very beneficial effect on the water quality of the receiving streams and lakes. Estimates and calculations of this nature can be made by using a joint EPA-USDA document entitled "Animal Waste Utilization on Cropland and Pastureland."

SOLUTION TO THE PROBLEM

The best way to solve any water pollution problem is not to allow the pollutant to reach the water in the first place. This has been the goal of the effluent guidelines developed several years ago under P.L. 92-500 which require zero discharge from confined animal production facilities unless a certain storm magnitude and frequency is exceeded. This regulation does not allow the producer in the U.S. to discharge effluent from *any* type of treatment system for animal wastes. The only options open to the animal industry then are to use the wastes in some manner or totally destroy them.

There are many ways to use animal wastes, in the production of fuel or feed, or as a fertilizer. Fuel and feed production require a technology not commonly found on the average farm, and, therefore, are not widely used today. Also, some of the processes used will produce some type of effluent that must be managed under the same zero discharge guidelines. Putting animal waste back on the land as fertilizer seems to be the only real answer to the problem.

Man has long recognized the beneficial effects of animal manures on crop growth and soil condition. Applying manure to the land completes the natural cycle of growth, death, and decay on land where crops are produced. The land contains legions of organisms capable of decomposing organic wastes of plants and animals into useful humus and the various elements essential for continued crop production. The application of animal wastes to the land sounds very simple, easy, and environmentally sound, but as with many other things, there are right and wrong ways to accomplish this task. Improper use or application of animal wastes to the land provides a potential pollution hazard to both surface and ground water. Incorrectly designed or managed manure collection and storage systems may lead to direct water pollution and can reduce the value of the animal wastes prior to delivery to the field.

The term best management practice (BMP) is used in the nonpoint source planning and implementation programs. Many take this to mean that for animal production there is a short list of practices that can be used to solve the problem. This is not quite correct. In fact, there are very long lists of components from which to choose to build a complete animal waste management system for a given set of conditions. However, the conditions will vary from one region to another and from one farm to another, even though the farms may be located side by side. What we have for BMP's then is a list of management system concepts which can be selected for a given area, and then we have a long list of ideas, components, and existing facilities which the farm planner may select and modify to finally design a true BMP for a given farm.

Many factors must be considered in selecting a BMP; only a few will be mentioned here to give some insight into the complexity of the system. Starting with the production unit, the number of head on the farm must be known to establish the daily volume of wastes to be considered. The form of the waste must be known, either liquid, slurry, or solid or some combination of each. The structural needs for the system may include runoff control ponds, manure storage areas, or manure pits below production buildings.

If storage is for slurry or liquid and odors become a problem, mechanical equipment may be added to control the odors. The size of the storage will depend upon number and size of animal and the length of storage time. Storage time is dictated by how often the land will be suitable for application due to weather and cropping patterns.

The cost of each proposed management system must also be evaluated, for it does no good to design a system that cannot be implemented because of economics. Many systems and their components are evaluated in a document developed by Ohio State University for EPA entitled "A Manual on Evaluation and Economic Analysis of Livestock Waste Management Systems." Computer models for sizing runoff control systems and applying the runoff to cropland have been developed by Kansas State University and Oregon State University. Also, the Cooperative Extension Service in each State has many fact sheets and other information that can be very useful to the developer of BMP's for animal waste problems.

The problem posed by pasturing of animals is different in that it is truly a nonpoint source problem, and the approach must be different from that used for confined animals. The pastured animals spread their own waste back on the grassland which produced the feed originally. They do not, however, spread it evenly. The wastes accumulate near watering locations and in shady areas. These areas are usually located very near stream banks and therein lies the potential problem. Some States have laws which require the fencing of streams which feed directly into public water supplies. This is a very positive way of approaching the problem.

There are other solutions: Shade can be provided away from streams; water tanks can be located near the new shade; and the trees could be removed from along the streams. There seems to be some correlation between the amount of forage remaining on the pasture and the quality of the runoff from the pasture. The more forage remaining, the better the quality of the runoff from that area. This possible correlation indicates that good forage and cattle management may provide the necessary BMP's for most of the nonpoint source (NPS) pasture problems.

The problem of bacterial pollution is one that will always be of concern in the NPS programs. Coliform counts from pastures have been reported to exceed stream standards at most locations. Several studies on water quality from grassland with and without animals indicate that bacterial pollution counts vary greatly and can be very high from the ungrazed grassland. This indicates that domestic animals are not the only problem; removing them from the areas may only allow wildlife numbers to increase. A great deal remains to be learned about the effectiveness of BMP's for unconfined animal production.

DECIDING ON BEST MANAGEMENT PRACTICES

Agencies or individuals charged with developing programs to control pollution resulting from animal production will need to use a systematic procedure to be able to properly identify NPS problems and to recommend practical BMP's to solve the problems. The most effective way of developing these local NPS programs will be to bring together a group of specialists to identify the problems and develop specific guidelines for the localized area. The responsible agency should seek assistance from all Federal, State, and university agencies active in the area. The group should include agronomists, soil scientists, hydrologists, economists, engineers, biologists, and most importantly, farmers who know the local area. These representatives working as a group and using all of the available information on animal waste management should be able to pinpoint problem areas, develop realtistic BMP concepts, and design evaluation programs that will be suitable for implementation and will solve the problem.

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USDA SOIL CONSERVATION SERVICE STANDARDS FOR LIVESTOCK MANURE MANAGEMENT PRACTICES

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ABSTRACT

Studies of nonpoint sources of pollution throughout the United States indicate that livestock and poultry operations often significantly reduce water quality in surface waters to which they drain. This can be in the form of excess nutrients, reduced dissolved oxygen, or increased coliform bacteria. The Soil Conservation Service (SCS) has developed standards for planning, designing, constructing, and operating several practices for managing manure to minimize pollution of surface and ground waters. The first step is developing an overall waste management system plan. Known as Practice 312, Waste Management System, it sets forth all the system components needed on a farm or ranch to properly manage manure from the time it is excreted to its ultimate use as a beneficial resource — usually for improvement of soil tilth and fertility. The overall waste management system may logically include other soil and water conservation practices to prevent degradation of water, air, soil, or plants. Minimum standards for these practices are set forth in the SCS National Handbook of Conservation Practices. These standards are supplemented by States as necessary to meet local requirements. A waste management system requires not only careful planning initially, but careful, consistent operation and maintenance by the owner.

INTRODUCTION

The Soil Conservation Service is an agency within the U.S. Department of Agriculture. We provide technical assistance to landowners through locally organized soil and water conservation districts to protect soil and water for long-term production. This technical assistance includes planning, designing, and supervising installation of systems for managing livestock and poultry manure.

Waste management systems serve two main purposes — efficient management of manure for beneficial use, and control of pollution. The need for manure management systems became apparent as livestock production under confined conditions increased in the late 1960's and early 70's. SCS modified existing soil and water conservation practices and developed new ones to meet this need.

Our first systems were simple in concept. They were aimed at controlling pollution from confined livestock areas. Typically, they included clean water diversions to prevent such water from reaching livestock areas, polluted runoff diversions or collection ditches to intercept flow before it reaches surface waters holding ponds to retain polluted runoff, and irrigation or other equipment to apply collected liquids on available land.

Systems now include waste storage structures, filter strips, and designation of waste utilization areas to provide more efficient management of the manure and runoff water and make beneficial use of contained nutrients.

SCOPE OF PROBLEM

While SCS provides technical assistance in installing about 3,500 waste management systems a year, this does not meet the need for such systems across the country. There are some 1,800,000 farms in the United States with some type of livestock or poultry. Possibly 800,000 of these can be considered confined operations.

Estimates of the number and size of confined feeding operations having a potential for polluting surface waters were made in 1976 by USDA and State research and extension personnel in major livestock-producing States. The estimates were made for 18 States representing 95 percent of beef production, 15 States with 90 percent of swine production, and 24 States with 85 percent of dairy production. The animal waste subcommittee of the USDA Environmental Quality Committee summarized the estimates and provided them to the U.S. Environmental Protection Agency to help in determining the impact of proposed feedlot regulations.

About 94,500 operations in the major livestock producing States pose a potential pollution threat because of discharge in manmade waste conveyances to surface waters, in a watercourse traversing the operation, or in operations with 1,000 animal units or more with a discharge reaching surface waters from a storm of less magnitude than a 24-year-frequency, 24-hour storm event. About 14,000 beef, 32,000 dairy, and 48,500 swine operations make up this total and represent 20, 19, and 11 percent of total production, respectively.

In addtion, approximately 105,000 operations with less than 1,000 animal units have discharges reaching surface waters from storms of less than a 25-year-frequency, 24-hour magnitude. This includes about 23,000 beef, 29,500 dairy, and 54,500 swine operations and 8, 16, and 11 percent of the total production, respectively.

The estimated 200,000 confined livestock operations with potential pollution problems represented 28 percent of the 719,000 total operations in the major production States.

POINT AND NONPOINT SOURCES OF POLLUTION

SCS is not a regulatory agency. It is our job to provide technical assistance to land owners and operators to help them comply with local State and Federal regulations. When a livestock or poultry operation is relatively large or is found to be a significant source of pollution, it is designated as a "point source" by the U.S. Environmental Protection Agency or the State regulatory agency. As a point source, the owner or operator must comply with the National Pollutant Discharge Elimination System. An NPDES permit is needed, and compliance requires that pollutants not be discharged to surface waters. This requirement means that all runoff from storms up to a 25-year, 24-hour event must be retained on the farm. While EPA regulations are uniform across the country, State regulatory agency designations of concentrated animal point sources vary considerably.

Smaller livestock and poultry operations are generally considered "nonpoint sources" of pollution. Control of such nonpoint sources involves the use of best management practices (BMP's). Depending on specific site conditions, BMP's can range from the no-discharge systems used with point sources to simply directing any discharge through grass filter areas. Even for the smaller operations, however, SCS generally recommends installing a livestock waste management system that prevents discharge of polluted water to surface waters.

LIVESTOCK WASTE MANAGEMENT SYSTEMS

SCS standards require that a complete waste management system be planned before individual practices are installed. This is to prevent the owner from investing in a component that may not be a logical part of a total system needed for that particular enterprise. Our concept of a complete waste management system is to provide facilities for management of manure from its production to utilization — usually on the land. While national standards must be met, practice standards may vary from State to State to recognize differences in climate, State and local regulations, and types of livestock and poultry operations. National practice standards are revised periodically on the basis of improved technology and experience. SCS national practice standards for a waste management system and its various components are summarized as follows.

PRACTICE 312 — WASTE MANAGEMENT SYSTEM

This practice comes before all others. It evaluates all liquid and solid waste sources on a farm and develops a complete system including all necessary components to manage them without degrading air, soil, or water resources. The practice considers the waste from the time of production to its ultimate use on the land. This practice determines if there is sufficient land to utilize contained nutrients in the waste and if the land is available at times compatible with crop management and labor requirements. While we emphasize management of wastes in a manner that conserves nutrients, there are situations where sufficient land is not available to utilize the nutrients. In such cases it may be best to plan practices where nitrogen loss is maximized.

A waste management system may consist of a single practice such as a clean water diversion, or a combination of several practices. It is important that individual practices be installed in a sequence that insures that each will function as intended without being hazardous to others. For example, a lagoon or holding pond should not be installed until planned diversion of outside sources of runoff has been accomplished.

Components of complete waste management systems may include, but are not limited to, the following practices: Debris basins, dikes, diversions, fencing, filter strips, grassed waterways, irrigation systems, pond sealing or lining, subsurface drains, waste storage ponds, waste storage structures, waste treatment lagoons, and waste utilization.

Another important element of a complete waste management plan is guidance for operation and maintenance. An operation plan is prepared for the owner, providing specific details for operation of each component of the system. Typically, such a plan should include:

1. Timing, rates, volumes, and locations for application of waste and, if appropriate, approximate number of trips for hauling equipment and an estimate of the time required.
2. Minimum and maximum operation levels for storage and treatment practices and other operations specific to the practice, such as estimated frequency of solids removal.
3. Safety warnings, particularly where there is danger of drowning or exposure to poisonous or explosive gases.
4. Maintenance requirements for each of the practices.

PRACTICE 425 — WASTE STORAGE POND

Waste storage ponds are used to temporarily store liquid and solid wastes, wastewater, and polluted runoff until it can be safely applied to land or otherwise used without polluting surface or ground water. They are constructed of earth and may have paved entrance ramps and bottoms to facilitate emptying.

A common use of waste storage ponds is to store polluted runoff from concentrated livestock areas such as feedlots and barnyards. Another common use is for storage of manure and milkhouse or milking center wastes.

Diversions or dikes are usually used in conjunction with waste storage ponds. Some divert clean water away from concentrated livestock areas, and others are designed to collect polluted runoff and direct it to the storage pond. They are designed to handle the same storm event which governs the storage pond design, usually the runoff from a 25-year, 24-hour rainfall.

Design of the waste storage pond must consider the maximum period between emptying. This varies according to climate, crops, and labor. The design volume must equal or exceed the total of the following:

With drainage area —	**Without drainage area —**
1. Manure, wastewater, and normal runoff[1]	1. Manure and waste water[1]
2. Normal precipitation less evaporation on pond surface[1]	2. Normal precipitation less evaporation on pond surface[1]
3. 25-year, 24-hour runoff	3. 25-year, 24-hour precipitation on pond surface
4. Solids accumulation[2]	4. Solids accumulation[2]

[1] Accumulated during the storage period.

[2] For the period between solids removal. This applies mainly to ponds used to store wastewater and polluted runoff and refers to the residual solids after the liquids have been removed.

Additional storage is often provided to meet management goals or State regulations. The most common comment heard from owners is that, if they were to construct a pond again, they would provide more storage to allow more flexibility in applying the manure or wastewater to the land.

When storing polluted runoff or liquid wastes, it is advisable to direct the waste through some type of solids removal device. This cuts down on the frequency of removing accumulated solids from such ponds and reduces problems when spray irrigation type equipment is used to apply liquids to the land. Common solids removal facilities include debris (settling) basins, low-gradient channels, and vegetative filter strips. Various mechanical devices are also available for this purpose.

There is some concern that waste storage ponds will pollute ground water. Certain gravelly soils and shallow soils over fractured or cavernous rocks should be avoided unless special precautions are taken. Research indicates and our experience shows that waste storage ponds rapidly seal in all except very coarse solids if the waste is organic and solids content exceeds about 0.5 percent. This seal is a result of settling of fine solids and biological action at the interface of the soil with the waste. It is good practice to retain some liquids in the ponds to maintain the seal. Extended drying breaks down the biological seal. The seal is reestablished when wastes are again introduced.

PRACTICE 359 — WASTE TREATMENT LAGOON

When animal or other agricultural wastes must be treated, waste treatment lagoons may be a logical component of a waste management system. Treatment may be needed for odor control or, where land for application is limited, to reduce nitrogen content of the waste.

Waste treatment lagoons are designed as anaerobic, aerobic, or aerated lagoons. For livestock wastes, they are often used to biologically treat milkhouse or milking center wastes, liquid manure from flush systems, or other types of liquid wastes.

Anaerobic lagoons are the most common type for livestock wastes. They require much less area and volume than aerobic lagoons and do not require the energy input of aerated lagoons. Volatilization of ammonia causes substantial loss of nitrogen from anaerobic lagoons, often allowing use of smaller land areas for the waste. When the lagoons are properly designed and managed, odors are not usually a problem in agricultural areas. However, when the lagoons are overloaded and often when they are being emptied, odors may be objectionable in residential areas. The effluent from anaerobic lagoons is not of sufficiently high quality for discharge to surface waters.

Aerobic lagoons are sometimes used for milkhouse or milking center wastes and for other relatively weak agricultural wastes. Because of large surface areas involved, they are not often used for treating liquid manure. Rarely is the effluent from aerobic lagoons of sufficient quality for discharge to surface waters.

Aerated lagoons are used primarily for odor control. The cost of the energy required generally prohibits their use for complete mixing and treatment of strong agricultural wastes. When used for odor control, they are designed to aerate the surface of the lagoon — the remainder is anaerobic. Once again, the effluent should not be discharged to surface waters.

Anaerobic lagoons are designed on the basis of volatile solids (VS) loadings ranging from about 3 pounds VS in the north to 7 pounds VS in the south per 1,000 cubic feet per day. Aerobic lagoons are designed on the basis of 5-day biochemical oxygen demand (BOD5) ranging from about 20 pounds BOD5 in the north to 60 pounds BOD5 in the south per acre of surface area per day. Aerobic lagoons treating animal wastes with a high chemical oxygen demand to BOD5 ratio often are aerobic only near the surface. Virtually all aerobic lagoons treating organic agricultural wastes have an anaerobic zone at the liquid-soil interface.

Waste treatment lagoons are not designed to treat polluted runoff. The irregularity of runoff events and variability of pollution loading are not amenable to rational design. Uncontrolled outside runoff is excluded from lagoons for these reasons.

PRACTICE 313 — WASTE STORAGE STRUCTURE

Waste storage structures include storage tanks and manure stacking facilities. In contrast to waste storage ponds, they are made of materials such as reinforced concrete, coated steel, wood, and masonry. As components of waste management systems, waste storage structures are used to temporarily store liquid or solid wastes until they can be safely applied on the land or otherwise used. Storage tanks are used for liquid and slurry wastes. Stacking facilities are used for wastes that behave as solids.

Waste storage structures are sized to store accumulated wastes, bedding, wastewater, and any needed dilution water for the maximum period that such wastes cannot be beneficially used. The period of cold and rainy weather often dictates how long wastes must be stored; however, stage of crop growth or availability of labor may influence length of storage. Structures are designed to insure that they are sound and of durable materials commensurate with the required service life, cost, and maintenance. Anticipated service life is broken into three categories — 10 to 20 years, 20 to 50 years, and over 50 years — depending on the specific enterprise and the owner's desires.

Provisions, such as entrance ramps and pumping and agitating ports, are provided for emptying waste storage structures. The owner or operator must have equipment available for his use in filling and emptying the structures, and he must provide warning signs, ladders, ropes, rails, and other devices as necessary for the safety of human beings and livestock. Proper ventilation of enclosed structures is a critical concern to prevent accumulation of explosive and toxic gases.

While waste storage structures are expensive components of waste management systems, they offer many advantages over waste storage ponds. Advantages include preservation of nutrient content of stored wastes, minimization of odors, management flexibility, and improved aesthetics. Occasionally, State regulatory agencies prefer or require waste storage structures rather than waste storage ponds.

PRACTICE 393 — FILTER STRIP

Filter strips have a definite place as components of waste management systems. While they have been used for years to filter sediment from water flowing from cropland, their use as a formal practice in waste management is relatively new. Their purpose is to remove sediment and other pollutants from runoff by filtration, infiltration, absorption, adsorption, decomposition, and volatilization.

Filter strips can be considered a useful and relatively inexpensive practice for reducing sediment and other nonpoint pollutants. To date, national design criteria do not spell out the limits of the effectiveness of filter strips. They are currently being used between feeding areas and streams for livestock on pasture, between areas where wastes are stored and surface waters, below feedlot areas to filter solids from polluted runoff before runoff is directed to holding ponds, and to a limited extent as facilities for reducing pollutants in runoff from concentrated livestock areas. It is hoped that additional research and experience will provide improved guidelines relative to length of filter area and reduction of various pollutants.

PRACTICE 633 — WASTE UTILIZATION

The purpose of the waste utilization practice is to safely use wastes to provide fertility for crop, forage, or fiber production; to improve or maintain soil structure; to prevent erosion; to produce energy; and to safeguard water resources. It completes a waste management system.

With most animal waste management systems, waste utilization refers to where and when manure should be applied to land. This practice is developed to match conditions in each State. Available nutrients are determined based on type and number of livestock and nitrogen losses related to method of management. For example, manure stored on an open lot and spread annually may lose up to 70 percent of its nitrogen whereas manure stored in a deep tank and incorporated into the soil before drying may lose only 20 percent.

Land areas available for application of manure and crops to be grown are determined. Amounts of manure to be applied are based on nutrient content of the manure and nutrient needs of the crop. Timing of applications is based on stage of crop growth and availability of labor and equipment.

Many other factors are considered in planning a waste utilization practice. They include rates of release of nutrients from manure, soil types, climate, and moisture need. If a lot of land is available, it may be best to apply only enough manure to meet phosphorus need and apply supplemental commercial nitrogen. If land is limited, it may be best to supply all the crop's nitrogen needs with manure. The most important factor is the owner's preference.

CONCLUSION

The U.S. Department of Agriculture is emphasizing greater use of organic wastes to improve soil tilth and fertility. Working with farmers and ranchers across the country, SCS is planning, designing, and supervising installation of waste management systems to abate pollution and to use organic wastes as resources.

We have discussed the general content of waste management practices as developed at the national level. Minimum standards are set forth in the SCS National Handbook of Conservation Practices. The SCS staff in each State supplements these standards as necessary to meet local conditions.

The key to a successful waste management system is the owner or operator. If he considers management of waste a nuisance and an unpleasant chore, the best conceived system just will not work. However, if he has a real interest in making beneficial use of the wastes, he will make his system work regardless of any shortcomings. A waste management system and its components require careful operation and maintenance. If a lagoon or waste storage pond or tank is not agitated and wastes removed on a regular basis, the job becomes much more difficult. Once weeds and brush with their massive root systems begin to form on the floating mat of a dairy manure storage pond, for example, it becomes virtually impossible to agitate and empty the facility.

SCS is emphasizing that complete waste management systems be planned before individual components are installed, that the plan include designation of areas for beneficial use of the waste, and that a written plan for operation of the system be developed with the land owner or operator.

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AGRICULTURAL NONPOINT SOURCE CONTROL OF PHOSPHORUS AS A REMEDY TO EUTROPHICATION OF A DRINKING WATER SUPPLY

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ABSTRACT

The water quality goal of the West Branch Delaware River Model Implementation Program, a U.S. EPA-USDA sponsored agricultural and silvicultural nonpoint source remedial program, is the deceleration of eutrophication in the Cannonsville Reservoir, a New York City drinking water supply. The river, which provides 75 percent of the reservoir's water, drains a 85,000 hectare forest (70 percent) and dairy agriculture (22 percent) watershed. Data collected from the reservoir by various investigators during 1971-1975 document the reservoir as a eutrophic system largely controlled by phosphorus. Total phosphorus (TP) loading estimates made from 1972-1975 data range from 39 to 117 tons/year. Based upon 1976-1978 data (n = 70),TP loading to the reservoir has been reduced to approximately 22 tons/year, largely because of improvements in point source discharges. TP loading remains near the level considered dangerous with respect to eutrophication. The program has directed approximately 75 percent of its Federal cost-sharing funds toward animal waste management practices, primarily barnyard runoff controls (275 dairy farms generate about 200 tons of phosphorus annually). A research and monitoring program will evaluate the effectiveness of barnyard runoff controls and the impact of the program on phosphorus export from the watershed and eutrophication in the reservoir.

INTRODUCTION

Increased attention is being given to the section 208 (P.L. 92-500) requirement calling for the development of processes to control agricultural and silvicultural nonpoint sources of water pollution. Responding to this requirement, the U.S. Department of Agriculture and the U.S. Environmental Protection Agency initiated the USDA/USEPA Model Implementation Program. The program employs existing agency funds to provide for land management in a select number of model watersheds to improve water quality.

In late 1977 the New York State USDA section 208 advisory committee nominated the West Branch of the Delaware River as a candidate for a Model Implementation Program, citing use impairment of the Cannonsville Reservoir as a New York City drinking water supply as the ultimate target. EPA had estimated that the reservoir was receiving over three times the phosphorus loading considered dangerous with respect to eutrophication. Nonpoint source phosphorus derived from cropland and animal wastes was suspected to account for a significant portion of the total load.

A technical conference convened in early 1978 after approval of the WBDR-MIP application provided the following conclusions to guide the MIP (N.Y. Dep. Environ. Conserv. 1978):

1. Nutrient (phosphorus) enrichment of the Cannonsville Reservoir is the critical water quality problem to be addressed.
2. Nonpoint source control measures with the greatest potential for reducing phosphorus loadings must receive the highest priority.
3. Measures relating to dissolved phosphorus must receive higher priority than those relating primarily to total phosphorus.
4. Measures not having a substantial relationship to the phosphorus loading, as defined in the guidance document, must receive a low priority.

The conference defined milkhouse wastes, barnyards, manure storage, and manure spreading as high priority sources of phosphorus, particularly dissolved phosphorus. Cropland, initially targeted for a large share of attention, was considered a low priority source because of its relatively unknown potential for phosphorus loading. The conference concluded that owing to the complex nature of the reservoir and the contribution of point sources, measurable improvement in the water quality of the Cannonsville Reservoir would not be expected during the program's 3-year period or shortly thereafter. The guidance document expressed the need for a baseline analysis of existing information on the river and the Cannonsville Reservoir.

This paper summarizes the available data for total phosphorus from previous investigations and attempts to provide a clearer perspective of the water quality goals of the program. While dissolved phosphorous is probably more relevant to eutrophication, the paucity of reliable dissolved phosphorus data for the WBDR prohibits its use to analyze trends in dissolved phosphorus loading.

STUDY AREA

The West Branch of the Delaware River watershed, located in the southeastern portion of New York State, is approximately 90 kilometers in length and drains in a southwesterly direction into the Cannonsville Reservoir (Figure 1). The reservoir is owned and operated as a public water supply by the City of New York. Land uses in the 116,000 hectare watershed (Table 1) are predominantly woodland (70 percent) and dairy agriculture (22 percent). Urban areas occupy only 1 percent of the study area. The population of the watershed is approximately 17,000. Watershed topography is rolling to mountainous. The average watershed slope is moderately steep (approximately 20 percent).

Climate is humid continental with temperatures averaging 8°C annually and precipitation averaging over 100 centimeters annually. Stream runoff from this precipitation averages 64 centimeters annually. The ground generally is frozen in Delaware County from December to March.

The Cannonsville Reservoir, formed by damming the West Branch of the Delaware River near Stilesville, N.Y., began filling in 1963. The surface area of the reservoir at a useful storage capacity (0.362 km^3) is 19.43 square kilometers and its mean depth is 18.6 meters. Seventy-eight percent of the reservoir watershed is drained by the river. The remaining 22 percent is accounted for by a number of small tributaries and direct input. Flow in the river below the dam is maintained by a hypolimnetic discharge during periods when crest capacity is not exceeded. The annual average turnover time is 0.45 year. Normalized flow data for the reservoir (U.S. EPA, 1974) indicate 55 percent water replacement during the period February-May during a normal year based upon the assumption of complete mixing. In light of the morphometry of the long and narrow West Branch arm of the Cannonsville Reservoir, complete mixing is probably a weak assumption and indeed much of the reservoir's water could be replaced during high flow by nondispersive advection.

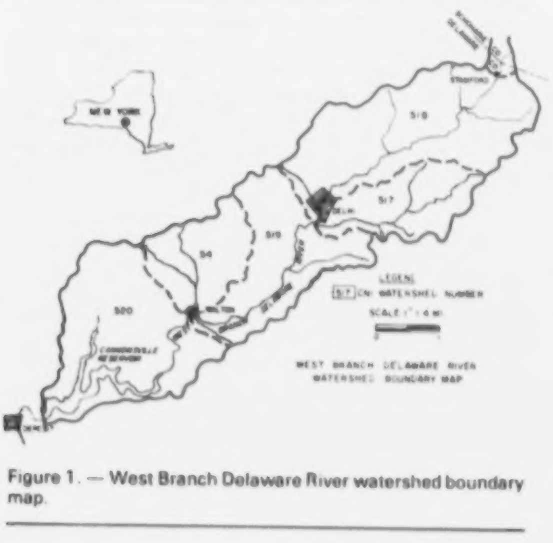

Figure 1. — West Branch Delaware River watershed boundary map.

Table 1. — Land use in the Cannonsville Reservoir watershed (Soil Conserv. Serv., 1977)

	Cannonsville Reservoir	
	ha	Percent*
Woodland	81,160	70
Cropland	16,318	14
Pastureland	8,891	8
Former cropland	3,342	3
Urban	1,013	1
Other	5,515	5
	116,239	

*Figures Total > 100 due to rounding.

RESERVOIR EUTROPHICATION AND PHOSPHORUS LOADING

The available water quality data for the reservoir are from four sources: a phytoplankton survey of the Delaware River Basin by Schumacher and Wager (1973), the National Eutrophication Survey (U.S. EPA, 1974), a limnological study of the reservoir by Wood, (1979), and the New York City Department of Water Resources routine water quality monitoring data. The U.S. Geological Survey maintained water quality stations at Walton and at Beerston where the river enters the reservoir and at Deposit (Figure 1) below the reservoir from May 1973 through April 1975. All concluded the reservoir was eutrophic.

Using an analysis developed by Vollenweider (1975), U.S. EPA (1974) estimated that the reservoir would remain eutrophic if annual surface loading exceeded 1.05 g P/m^2/year. If loading was reduced below this limit, the reservoir could eventually become mesotrophic. EPA (1974), employing 1972-1973 data, estimated surface loading to be 4.26 g P/m^2 year. In this estimate the river accounted for 97 percent of the load. Wood (1979), using 1974-1975 data estimated surface loading to be 1.37 to 1.66 g P/m^2 year. The six estimates of TP loading to the Cannonsville Reservoir made prior to 1980 range from 27 to 117 tons/year. In these estimates the river accounted for a minimum of 77 percent of the load. Known point sources accounted for 27 to 41 tons/year. An estimate of 23 tons/year was generally employed for a cheese processing plant in Walton.

Much of the variability in the estimates can be accounted for by the assumptions concerning stream discharge and sample collection. While some investigators employed long-term average or normal flow, others considered flow for a specific year. The EPA (1974) and Geological Survey (1974, 1975, 1976) data were collected from November 1972 to April 1975. All of the pre-1980 loading estimates with the exception of Wood's (1979) have used the combined EPA and Survey data (n = 36). Wood's (1979) estimate is the only one incorporating a substantial amount of data from 1975. His substantially lower estimate probably reflects the reduction in TP loading resulting from the phaseout in the use of phosphorus-based cleaners at the cheese plant in Walton during 1975. Brown and

Rafferty (1980) examined estimates of phosphorus loading to the reservoir in greater detail and addressed the limitations of the 1972-1975 data. They concluded that the sampling program for the river below Walton may have biased results due to diurnal and weekly variations in the cheese plant's effluent quality. Such a bias could have resulted in overestimating the annual load to the reservoir.

A statistical summary of the entire TP data set for the river above Walton and at Beerston, where the river enters the reservoir, from 1973 to 1979 is presented in Figure 2. It is readily evident that the range of TP in the pre-1975 data is substantially greater for Beerston than for the river above Walton. Based upon 95 percent confidence limits as presented in Figure 2, the mean TP concentration at Beerston was significantly higher than above Walton during 1973-1974. Similarly, the TP concentration at Beerston during 1973-1974 was significantly higher than for the period 1974-1979. The dramatic reduction in TP concentration that occurred in the Beerston record in 1975 corresponds in time to a phaseout in the use of phosphorus-based cleaners at the cheese plant in Walton. Based upon incremental drainage area, stream flow at Beerston is generally 10 percent greater than above Walton. Incremental flow enters diffusely from a predominantly forested watershed.

Observations by Schumacher (pers. commun.) corroborate the reduction in nutrient loading that occurred in the Walton area during 1975. Schumacher and Wager (1973) had noted that unlike the river above Walton high concentrations of fungi were found in samples approximately 9 kilometers downstream from Walton. The fungi there were dense enough to prohibit phytoplankton counting. Schumacher observed that the fungi were no longer visible in the same area during and after 1975, and were replaced by filamentous green forms.

Log TP concentration on log flow linear and second order polynomial regressions performed on the 1976-1979 data yielded no significant relationships on which to base a loading estimate. Based upon the 1976-1978 mean TP concentration at Beerston, .023 mg/l, mean stream discharge for the same period, 25 m³/s and 95 percent confidence limits on the mean TP concentration results in an annual TP loading estimate of 19,300 ± 4,800 kg/year. Employing the EPA (1974) TP loading estimates from tributaries, immediate drainage, and direct precipitation to the reservoir, 2,700 kg/year, the total loading to the reservoir would be 22,000 kg/year or 1.14 g/m²/year. Using Vollenweider's (1975) model, EPA estimated that the eutrophic rate of phosphorus loading was 1.05 g/m².

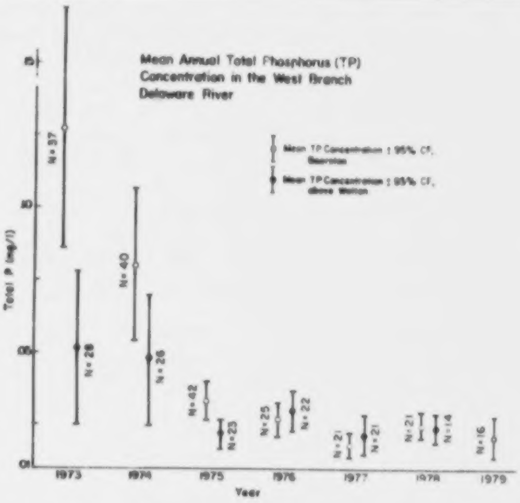

Figure 2. — Mean annual total phosphorus (TP) concentration in the West Branch Delaware River.

Table 2. — Summary of TP loading estimates to the Cannonsville Reservoir

Investigator	Data	Estimate metric tons/year	Methods
Hydroscience (1974)	USGS (1974, 1975) EPA (1974) Nov. '72 — Oct. '74	76 — 117	Product of mean TP concentration and mean flow for different periods of interest
EPA (1974)	EPA (1974) Nov. '72 — Oct. '73	83	Sum of products of normalized monthly flow and empirically adjusted TP concentration
Bricke (1975)	EPA (1974) USGS (1974, 1974)	39	Log [TP] on log flow linear regression to determine concentration for flow duration analysis.
Bricke (1975)	EPA (1975) USGS (1974, 1975)	49	Log [TP] on log flow 2° polynomial regression to determine concentration for flow duration analysis.
Goodale (1975)	EPA (1974) USGS (1974, 1975)	67	Product of mean annual flow and mean TP concentration.
Goodale (1975)	EPA (1974) USGS (1974, 1975)	40	Log [TP] on log flow linear regression to determine concentration for flow duration analysis.
Wood (1979)	Wood (1979)	27-32	Not reported.
Brown and Rafferty (1980)	NYC Dep. Water Resour. unpubl. data, 1976-1978	22	Product of mean TP concentration and mean flow for the same period (1 — 2 order polynomial regressions of log (TP) on log flow were not significant.)

INVENTORY OF PHOSPHORUS SOURCES IN THE CANNONSVILLE RESERVOIR WATERSHED

There are five municipal sewage treatment plants, a meat processing plant, and a hubcap plating plant that discharge directly to the river. The City of New York Department of Water Resources has been monitoring the quality of these effluents since 1977. Their data is summarized in Table 3. Known point sources contribute about 9,900 kg/year of TP. Approximately 75 percent of the point source load comes from the Walton sewage treatment plant, which has a mean TP effluent concentration of 8.0 mg/l.

In addition to point sources, EPA (1974) collected and analyzed 14 samples during 1972-1973 from each of three streams tributary to the reservoir draining forested land. The mean TP concentration in Dry Brook and Maxwell Brook was 0.015 mg/l. The third stream, Dryden Brook, had a mean TP concentration of 0.018 mg/l. Assuming unit area runoff from forested land equals that for other land uses, and employing the 1976-1978 mean stream discharge at Beerston (25.2 m^3/s), a TP concentration of 0.015 mg/l corresponds to a unit area TP load of 0.14 kg/ha/year. This load is well within the range reported for forested land in other areas (Uttormark, et al. 1974). Approximately 11,000 kg/y would be delivered to the Cannonsville Reservoir from forested land.

Remaining avenues of TP loading to the river and the reservoir include export from agricultural and silvicultural activities, urban runoff, septic systems, landfills, and direct precipitation. Estimates of these nonpoint sources are summarized in Table 3.

Logging operations appear to be minor contributions of phosphorus. Results of an aerial survey conducted by New York foresters during March 1979, indicated over 70 sites in the watershed had been logged since 1974. Forty of these sites were surveyed. The average size of the surveyed logging operations is 19.3 hectares of which logging roads on the sites averaged 0.7 hectares. Logging roads are generally the major sites of erosion on logging sites in the watershed; cleared areas typically have adequate soil cover (Trotta, 1980). Slavicek (1980) estimated that only eight of the 40 sites contributed sediment to streams. Unit area sediment loading rates developed for 12 transects, using the Universal Soil Loss Equation (Wischmeier and Smith, 1965), indicated sediment loading ranged from 4.29 to 247.0 tons/ha/year.

Agricultural sources of TP are related primarily to dairy farming although some beef and poultry farms are located in the basin. These sources include loading from barnyards, cropland and pastureland, and from handling of animal and milkhouse wastes. Because little data are available, estimates of TP loading from cropland, pastureland, and grassland in addition to urban runoff (Table 3) should be viewed in an order of magnitude sense.

Loading from barnyards, animal and milkhouse waste handling are more difficult to estimate, though their potential can be addressed. The 15,000 dairy cows and 5,000 replacements on a total of approximately 275 farms in the watershed (Soil Conserv. Serv., 1978) would be expected to produce approximately 300,000 tons of manure per year containing approximately 200 tons of phosphorus.

Table 3. — Inventory of phosphorus sources to Cannonsville Reservoir (Brown and Rafferty, 1980)

	River distance from reservoir km	Mean flow m^3/d	[TP] mg/l	Annual load kg/y
Point Sources				
Sewage treatment plants:				
*Stamford	72	2,498	1.25 (n = 50)	1,140
*Hobart	69	568	1.42 (n = 47)	294
*South Kortright	61	57	1.95 (n = 41)	41
*Delhi	35	1,287	2.00 (n = 51)	940
*Walton	8	2,498	8.00 (n = 14)	7,294
Industrial effluents:				
*Parnett	53	57	3.56 (n = 43)	74
*Delchrome	8	114	2.20 (n = 41)	92
		Area ha	Unit area load kg/ha·y	
Nonpoint Sources				
Land runoff:				
*Forest		81,160	0.14	11,362
Cropland		16,318	0.3	4,895
Pastureland		8,891	0.1	889
Grassland, other		8,857	0.1	886
Urban		1,013	1.1	1,114
Precipitation		1,942	1.02	1,981
		TP waste load kg/y	Delivery ratio	
Other Diffuse Sources				
Barnyards		12,000	?	
Milkhouse wastes		3,000-12,000	?	
Septic systems		1,750-3,500	?	
Total				31,002

*Data available from the watershed.

Based upon time spent in the barnyard by dairy cows (Mattern, pers. commun.), 12,000 kg/year of TP would be deposited on barnyards. Approximately 25 percent of these barnyards are adjacent to streams and 70 percent are within 67 meters of streams. Draper, et al. (1980) found that generally less than 10 percent of manure phosphorus is exported from open lots and employed a "best estimate" of 5 percent in analyzing phosphorus loading to the Great Lakes from livestock in the Ontario Basin.

While soils generally have a strong affinity for inorganic phosphorus, spreading manure on frozen ground has been shown to result in the export of up to 13 percent of manure phosphorus (Minshall, et al. 1970). In the West Branch watershed few farmers have manure storage facilities. Manure is spread daily during winter at locations often determined by accessibility. It should be noted that terrestrial systems conserve phosphorus, and only a small portion of that stored in undisturbed watersheds is exported (Hobbie and Likens, 1973; El-Baroudi, 1975). Klausner, et al. (1974) demonstrated that even combining high phosphorus fertilization rates (49 kg/ha/year) and poor soil management, resulted in export of less than 1 percent annually of inorganic phosphorus.

The major source of TP in milkhouse wastes is phosphorus-based cleaners used to clean bulk tanks and pipelines. These cleaners contain up to 8.7 percent phosphorus by weight. Milk, which contains approximately 1 g/l phosphorus, accounts for an insignificant portion of TP in milkhouse wastes due to the relatively small quantity wasted. Assuming 0.34 to 1.4 kg/day of 8.7 percent phosphorus cleaners are used on each of the 275 farms in the watershed, a general range of 3,000 to 12,000 kg/year of phosphorus is handled as milkhouse wastes. Part of these wastes enters septic tank-leach field systems; the remainder is discharged to dry wells, tile fields, or surface ditches.

DISCUSSION

Phosphorus loading to the Cannonsville Reservoir has been significantly reduced through managing point sources. However, preliminary monitoring data collected during summer 1980, indicate the reservoir is still eutrophic. The remaining point sources and nonpoint sources of phosphorus are summarized in Table 3. The most credible estimates are those for point source discharges and forest runoff, because of availability of data from the watershed. Other estimates should be viewed in an order of magnitude sense. The sum of individual source loading estimates for the river at Beerston is 24,500 kg/year (Brown and Rafferty, 1980) as compared to 19,300 ± 4,800 kg/year based upon New York City data for the river at Beerston. The conservative approach used in estimating nonpoint source loads where local data were unavailable probably contributes to the general agreement of the two estimates, considering the lack of high flow data in the New York City data. For example, while TP loading from cropland is probably underestimated, the rather short hydrologically active periods when the greatest percentage of phosphorus from cropland and other nonpoint sources would be exported were not well represented by the New York City monitoring.

Although the estimates of phosphorus loading from point sources to the river are generally good due to the availability of data, calculating a nonpoint source load by difference between the TP estimate made from data collected at Beerston is an especially tenuous procedure. It is likely that the TP estimate made from New York City data for the river at Beerston underestimates the true annual load. In addition, the assumption of conservative transport of sewage phosphorus inherent to the calculation may not be operationally valid for the river. Carlson, et al. (1978) demonstrated the attenuation of soluble phosphorus flux in streams below sewage treatment plants. The mechanism of attenuation was incorporation with bed sediments. If sewage phosphorus incorporation with bed material is a significant mechanism in the river, the delivery of this point source phosphorus to the reservoir would be synonomous with the major nonpoint source loads during those hydrologically active periods that received no special consideration in the New York City monitoring program.

With respect to eutrophication, that portion of the total phosphorus load which is available for algal growth is relevant. Lee, et al. (1980) suggest that available phosphorus from urban and rural runoff generally includes the soluble molybdate reactive phosphorus fraction and approximately 20 percent of the particulate phosphorus. Orthophosphorus accounted for approximately 60 percent of TP in samples collected from the river at Beerston by EPA (1974).

The larger portion of the phosphorus from point sources is probably available phosphorus. In streams tributary to the Cannonsville Reservoir draining forested land, approximately 50 percent of TP was present as orthophosphorus (EPA, 1974). Much of the phosphorus exported from cropland is likely to be sorbed to or precipitated on soil particles (Burwell, et al. 1977; Alberts, et al. 1978) and thus not immediately available. Much of the phosphorus loaded directly to the lake by precipitation could be unavailable (Lee, et al. 1980). Clearly, a better definition of phosphorus sources, particularly nonpoint sources and their bioavailability is needed.

The Walton sewage treatment plant appears to be a major contributor of phosphorus to the river. Considering the morphometry of the long and narrow river arm of the reservoir, the increase in phosphorus concentration caused by the Walton sewage treatment plant (.047 mg/l for normal July and August flow) probably insures the eutrophication of at least the upper reaches of the reservoir during summer months. It is more difficult to speculate on the plant's impact on the reservoir as a drinking water supply, since the drinking water tunnel inlets are located approximately 20 kilometers down the 27 kilometer-long reservoir.

While costs and effectiveness associated with point source improvements are well defined, the effectiveness of nonpoint source controls is poorly understood.

For agriculture, the New York Model Implementation Program is directed primarily toward controlling barnyard runoff. Of the 154 barnyards given priority based upon proximity to streams (<100 m), 90 farms participated in the program during 1978-1979; 67 signed up for installation of barnyard runoff controls. Of the $449,000 in Federal cost sharing funds

Table 4 — Practices used in the West Branch, Delaware River Model Implementation Program

Agricultural (Based upon data from farms where practices were installed during 1978-1979)

	Numbers of farms where applied	Average quantity per farm
Barnyard Runoff Controls		
Practice		
Roof gutters	27	36m
Tile	45	68 m
Drop inlet	11	1 Unit
Concrete pad & gravel	46	1 Unit
Fencing	36	150 m
Diversions	26	93 m
Waterway	9	0.3 m
Open ditch	7	48 m
Land grading	3	0.4 ha
Costs	50	$3,137
Other		
Temporary field storage of animal waste	1	
Animal storage facility	2	
Milkhouse waste filter strip	1	
Critical area planting	16	0.4 ha

	Silvicultural Number of participants	Total hectares
Activity (as of 9/79)		
Recon for Management Advice	34	114
Inspections	30	128
Management Plans		
Prepared	8	256
Revised	2	45
Timber Stand Improvement Marking	19	66
Sawtimber Marked	4	36
Pulpwood Marked	1	2.4
Fuelwood Harvested	1	5
Information & Education	54	
Log Road Erosion Control (6,450 m)		

committed during the first 2 years, approximately 75 percent is directed toward animal waste management practices: in particular, barnyard runoff controls to limit the size of barnyards, restrict direct access of cows to streams, divert surface flow from entering the barnyard, increase drying of barnyards, and facilitate collection of deposited manure. The controls consist of any combination of roof gutters and leaders, drop inlets, land grading, concrete pads and gravel, buffer strips, fencing, tile drainage, diversions, waterways, and open ditches (Table 4).

While the primary emphasis has been placed on barnyard runoff controls, a number of animal waste storage facilities, access roads to cropland, and a grassed filter strip for processing milkhouse wastes have been applied on farms in the watershed. The implementation of agricultural practices is being directed locally by the Delaware County Soil and Water Conservation District and the Soil Conservation Service in Walton.

The silvicultural activities are being administered by the U.S. Forest Service and the New York State Department of Environmental Conservation's Division of Lands and Forests. Their work has included an aerial reconnaissance of logging operations in the watershed and recruitment of landowners' and loggers' cooperation through a public information campaign. After the participating landowner's or logger's site is inspected, he can be advised of proper logging site management, have logging trails and timber marked by foresters, and possibly have a complete management plan prepared for the site (Table 4). In light of the increasing demand being placed upon wood as an energy source, growth in logging operations should coincide with erosion control practices to maintain logging's status as a minor contributor of phosphorus.

The evaluation of the water quality impact of the program resulting particularly from agricultural management practices, is the goal of a research and monitoring effort sponsored by the U.S. Environmental Protection Agency and conducted by investigators from the New York State Department of Environmental Conservation, the State College of Agriculture and Life Science, Cornell University, and the Soil Conservation Service. It could be reasonably assumed that the WBDR-MIP is targeting a less than 10 percent reduction in TP loading to the reservoir through the management of barnyard runoff. The evaluation will address three major processes:

1. Phosphorus loading to tributaries from barnyard runoff, manure spreading, and milkhouse wastes.
2. Phosphorus delivery to the Cannonsville Reservoir from the WBDR.
3. Eutrophication of the Cannonsville Reservoir.

These processes differ in the analytical sensitivity required to detect changes in them. The cumulative effect of the program on phosphorus loading to the Cannonsville Reservoir and its eutrophication may not be discernible in measurements taken at the reservoir because of the complexity of phosphorus sources and phosphorus transport and the relatively small target loading reduction. However, the ongoing research and monitoring of individual barnyard sources will permit at least an *a priori* assessment of the program's impact on the reservoir. In addition, the effectiveness of specific management practices in controlling nutrient export from barnyards will be evaluated.

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RESERVOIR PROTECTION BY IN-RIVER NUTRIENT REDUCTION

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ABSTRACT

If in the catchment area of a reservoir the portion of phosphorus compounds from diffuse sources prevails, phosphorus input can be reduced by chemical treatment of the main tributary. This scheme has been applied by Wahnbachtalsperrenverband for the oligotrophication of Wahnbach Reservoir (volume 40,000,000 m^3). At the point where River Wahnbach flows into the reservoir the incoming water taken from the pre-reservoir which serves as a reserve basin is treated by precipitation, flocculation with iron-III-salts at pH 6.4, and filtration. With this the total P-concentration is reduced by 99 percent to 4 $\mu g/l$ P as an average. Turbidity also is reduced to a residual of 0.05 FTU and dissolved organic carbon is reduced by 60 percent. This is achieved by energy-input controlled direct filtration (Wahnbach system) developed by Wahnbachtalsperrenverband. The treatment process includes precipitation of dissolved phosphates, destabilization of colloids and suspensoids, agglomeration of formed microfloc to large, well filtrable flocs and three layers' filtration with maximum 15 m/h filtration velocity. The maximum throughput of the plant amounts to 18,000 m^3/h. The 3 years' run of the plant shows, that by drastically reducing the annual average P-concentration from 100 $\mu g/l$ to 4 $\mu g/l$ the eutrophic Wahnbach Reservoir is transformed from the eutrophic to the oligotrophic-mesotrophic status. The annual average concentration of all tributaries including precipitation was reduced in 1979 for the first time to 16 $\mu g/l$, distinctly lower than the tolerable annual average concentration of 20 $\mu g/l$. At present, the dominating phosphorus load comes in via small marginal tributaries of the reservoir. This input will be reduced by further special measures.

INTRODUCTION

The origins of phosphorus in the catchment area of a reservoir are varied and detailed examinations have to be carried out to determine the most important phosphorus sources. One differentiates here between 'point' and 'diffuse' phosphorus sources. If diffuse phosphorus sources dominate in a catchment area, then there are only a few methods of reducing loading (Bernhardt, 1978).

One of these methods entails treating the whole of the inflowing main tributary using chemical processes to control the phosphorus input. This chemical treatment has been used on the Wahnbach Reservoir in the Federal German Republic for 3 years (Bernhardt, Clasen, and Schell, 1971; Bernhardt and Schell, 1979). It can be applied to those reservoirs which have only one or two tributaries or a gathering channel from a neighboring catchment area and in which phosphorus is a minimum factor.

EUTROPHICATION OF THE WAHNBACH RESERVOIR

Increased phosphorus input has gradually eutrophied the Wahnbach Reservoir (40,000,000 m^3 content) since impounding began in 1957. This eutrophication process made it more and more difficult to treat the raw water taken from the reservoir for drinking water. At the end of the 1960's and beginning of the 1970's, masses of blue-green algae *Oscillatoria rubescens* appeared. This not only colored the water but the *Oscillatoria* broke through the filter. Using a special flocculation process with a double dose of flocculant combined with a dose of polyelectrolyte, we generally mastered this calamity (Bernhardt and Clasen, 1973).

Sometimes the mass development of large diatoms such as *Melosira italica* or *Melosira islandica* caused shorter filter-run times because the rapid sand filter became blocked after 4 or 5 hours. Later, the small blue-green algae, e.g., *Coelosphaerium naegelianum* grew in increasing quantities and despite a 90 to 99 percent reduction could not be totally eliminated from the water because they were present in too large a concentration (20,000 to 200,000 cells/ml in the raw water). This was unsatisfactory for obtaining drinking water from the Wahnbach Reservoir.

Difficulties particularly arose every autumn during drinking water treatment as a result of algal organic compounds (Bernhardt and Wilhelms, 1978). They also disturbed the flocculation and disinfection and were partly precursors for the development of trihalomethane (Bernhardt and Hoyer, 1979). All these factors compelled the Wahnbach Reservoir Association to take steps to reduce the high input of phosphorus compounds into the reservoir from its catchment area so that the amount of bioproductivity in the reservoir would become tolerable again.

THE CONCEPTION OF THE OLIGOTROPHICATION OF THE WAHNBACH RESERVOIR

Experiments carried out over a period of several years showed that phosphorus compounds in the reservoir are the limiting factor (Figure 1). Every year orthophosphate was depleted in the reservoir down to a concentration of $<10\ \mu g/l$ owing to bioproductivity. However, concentrations of nitrogen and carbon hardly decreased at all. This meant that it was sufficient to reduce the phosphorus in the lake only so that algal development would be controlled.

Detailed studies on the origin of the phosphorus compounds from the catchment area of the Wahnbach Reservoir showed that more than 50 percent originated from diffuse sources. Only a small part of them came from locatable point sources (Bernhardt, et al. 1978). For this reason the Wahnbach Reservoir Association decided to erect a plant to decrease the phosphorus content in the main tributary flowing into the reservoir; this tributary transports 90 percent of the annual reservoir influent and approximately 90 percent of the phosphorus entering the reservoir. Figure 2 shows the site of this plant. After flowing into the pre-reservoir which serves as a floodwater retention basin, the water is pumped into the phosphorus elimination plant (PEP) and treated by precipitation, flocculation, and filtration which aims at reducing the total phosphorus content to $<10\ \mu g/l$ P. The water treated in this way then flows through a channel directly into the main reservoir. The construction of the phosphorus elimination plant (PEP) was completed at the end of 1977 and the plant has been in operation now for 3 years.

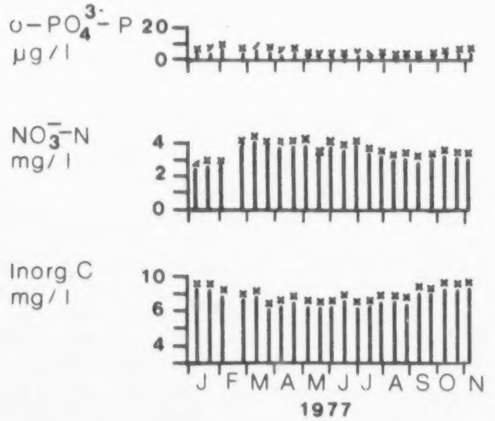

Figure 1. — Conc. of O-PO -P, NO -N and inorganic C calculated as mean values over the mixing depths in the Wahnbach Reservoir (April-Sept.: 0-10 m depth, at other times all depths).

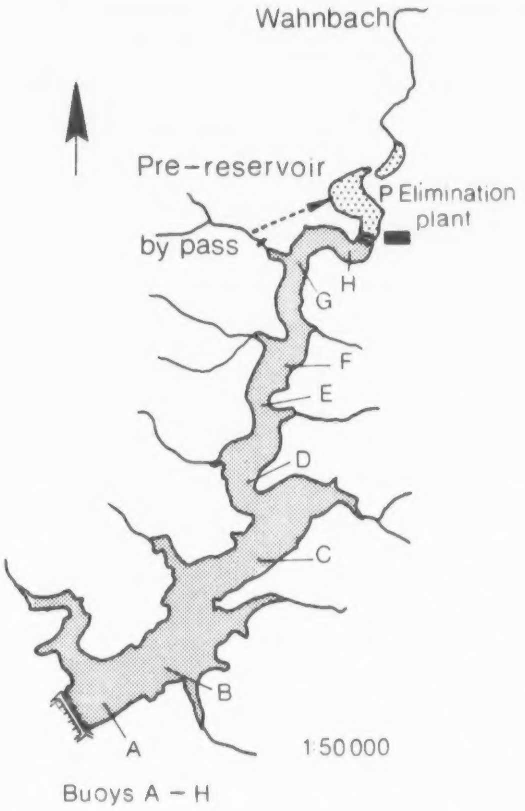

Figure 2. — P-Elimination plant on the Wahnbach Reservoir.

RESULTS OF PHOSPHORUS ELIMINATION AS REGARDS THE EXTENT OF NUTRIENT LOADING OF THE WAHNBACH RESERVOIR

Figure 3 shows the amount of phosphorus fed into the reservoir during 1977-1979 subdivided into individual nutrient sources. The black columns show the quantities of phosphorus (in monthly values) which the River Wahnbach transported into the reservoir before the phosphorus elimination plant was in operation. The columns marked with diagonal lines show the total phosphorus freight which was retained in the elimination plant after the plant was operating. This quantity of phosphorus would have been fed into the reservoir during 1978/79 if the plant had not been operating. The black columns in November 1977, May and June 1978 and March and December 1979 show the quantities of phosphorus which could not be treated owing to high water peaks (pre-reservoir overflow). The dotted columns are the quantities of P which were transported into the main reservoir based on the concentration of P still present in the plant outflow ($< 5 \mu g/l$ P).

Before the plant began operating, most of the phosphorus was transported into the reservoir by the River Wahnbach. The quantities of phosphorus which reached the reservoir via lateral inflowing tributaries and precipitation were insignificant. After the plant was put into operation, it happened that more phosphorus was transported into the reservoir via these lateral tributaries (white columns) than had been transported by the main tributary. Thus the phosphorus loading of the reservoir caused by the lateral streams became decisively important.

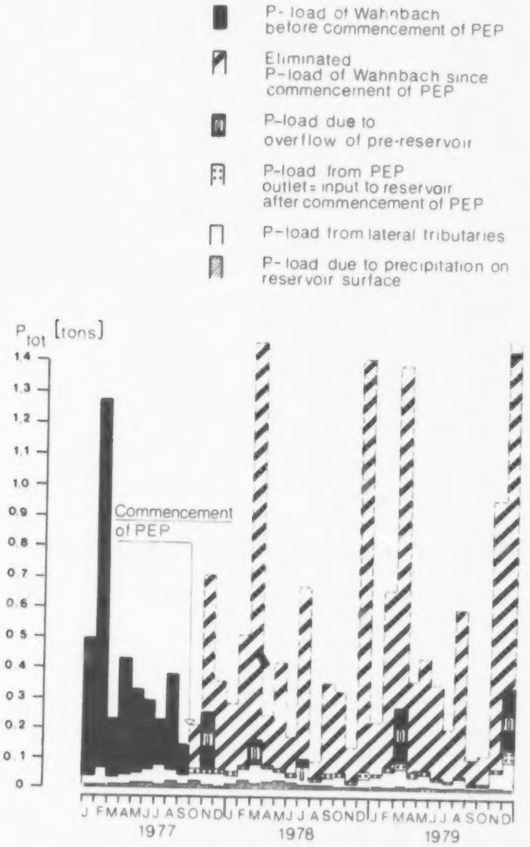

Figure 3. — Phosphorus load of Wahnbach Reservoir from different sources before and after commencement of phosphorus elimination plant.

THE AIM OF PHOSPHORUS REMOVAL AT THE MAIN TRIBUTARY

When the pilot plant for removing phosphorus was installed on the main tributary of the Wahnbach Reservoir, it was not precisely known how far the P-concentration in the main tributary had to be lowered to achieve a tolerable water quality in the reservoir. At that time one could only rely on Sawyer (1966), who had found out from practical experience that water quality caused no problems in those lakes in which the spring overturn P-concentration was below 10 to 15 μg/l P. To ensure that this concentration was attained in the Wahnbach Reservoir, it was decided to reduce the P-concentration in the main tributary to 10 μg/l P. Today we know that this concentration in the main reservoir tributary is still too high if the reservoir is to become oligotrophic to mesotrophic. If one applies Vollenweider's formula (1976) for estimating the tolerable P_{tot}-concentration in all the reservoir inflows $[\overline{PT}]_{i,c}$ onsidering the average P_{tot}-concentration in a reservoir $[\overline{PT}]_{\lambda} = 10\ \mu/l$ calculated over a period of 1 year,

$$[\overline{PT}]_{i,c} = 10\,(1 + \sqrt{\tau_w}\ )$$

one then obtains for $[\overline{PT}]_{i,c} \approx 20\ \mu/l$ if the retention time of the water in the reservoir $\tau_w = 1$ year.

Three years of operating the plant have shown that the P_{tot}-concentration of an average of 100 μg/l in the Wahnbach (60-180 μg/l P_{tot}) has been reduced to an average of 4 μg/l P_{tot} in the plant's outflow. This means that the P_{tot}-concentration of all the inflows into the reservoir including precipitation was reduced to 16-20 μg/l P_{tot}. This figure corresponds to the calculated P_{tot}-concentration of all the inflows.

PRINCIPLE OF THE PHOSPHORUS ELIMINATION PLANT

The phosphorus elimination plant (Figure 4) is designed for a maximal flow of 5 m³/sec. Thus the fivefold amount of the long-time average flow of the River Wahnbach (1 m³/sec) can be treated. Together with the storage capacity of the pre-reservoir which serves as a water retaining basin with a capacity of 500,000 m³, up to 8 m³/sec can be treated, at least for a limited period.

The phosphorus elimination plant should meet the following requirements:

1. It should run for several weeks on full capacity.
2. Rapid variation of flow capacity between 3,000 and 18,000 m³/h.
3. Operation for a few hours with intervals of several days, frequent switching on and off without decreasing quality of filtrate.
4. No drop of efficiency at water temperatures of 0 °C (winter running).
5. Treatment of water with high turbidity (up to 100 mg/l content of solids (105°C) without shortening the duration of filter runs to less than 10 hours at the maximal filtration rate of 16 m/h.
6. Decrease of the total phosphorus content to values ≦5 μg/l.
7. Treatment should be arranged in such a way that ≧ 99 percent of the plankton occurring during the summer months (max. 400,000 cells/ml) can be eliminated from the water. Algal cells which break through the filter cause high concentrations of undissolved and dissolved organically bound phosphorus compounds in the filtrate. This means an undesired phosphorus load in the reservoir.
8. Removing 99 percent of inorganic turbidity flushed into the reservoir after the erosion of arable land which is rich in phosphates.

9. Flocculation is carried out with 4-10 mg/l Fe^{+3}. The total iron concentration in the effluent should not exceed 50 µg/l Fe.

10. The plant should be constructed as compactly as possible to be economically and technically worthwhile. The filter area is 1,100 m^2. With a throughput of 4-5 m^3/sec. one has a filter velocity of 16 m/h.

To achieve all this we developed the energy-input controlled direct filtration called 'Wahnbach System' with the following steps (Figure 4):

1. Precipitation of the o-phosphate ions present in the water by adding iron-III-ions in the acid pH zone (pH 6.0-7.0, average pH 6.4).
2. Destabilization of the colloids and suspensoids in the raw water to which the precipitated iron phosphate products also belong. This is done by adding iron-III-ions as a flocculation agent.
3. Agglomeration by means of transport. The microflocs unite in a subsequent agglomeration step and form larger, partly visible flocs. By adding a cationic polyelectrolyte they are made suitable for filtering. The type of cationic polyelectrolyte used changes according to the time of the year.

Required for both destabilization and agglomeration, the amount of special energy-input is adapted to the quality of the raw water and the throughput. The filter of the phosphorus elimination plant consists of three layers of various granulations and densities:

1. Upper layer — 30 cm active carbon, granulation 3-5 mm, effective grain size 3.9 mm.
2. Middle layer — 125 cm hydro anthracite, granulation 1.5-2.5 mm, effective grain size 1.7 mm.
3. Bottom layer — 50 cm quartz sand, granulation 0.7-1.2 mm, effective grain size 0.9 mm.

EFFICIENCY OF THE PHOSPHORUS ELIMINATION PLANT

The quality of the filtrate is illustrated by Table 1. The average and the minimum and maximum values and the average elimination are shown. About 95 to 99 percent of the phosphorus compounds are eliminated. The total phosphorus concentrations in the filtrate amount to 4 µg/l 1 on a 2-year average. They are 60 percent less than the P-concentration we aimed at achieving.

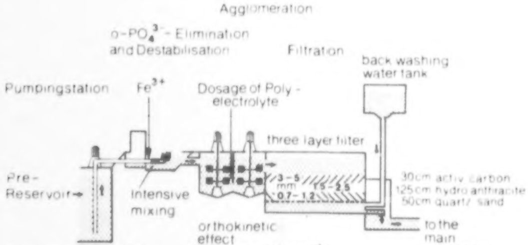

Figure 4. — Principle of the t-filtration with controlled energy input, 'Wahnbach System'.

The decrease of soluble organic compounds varies depending on the sum parameter chosen for characterization. We have 58 percent elimination of dissolved organic carbon because there is only 25 percent elimination of the organic compounds with a molecular weight of <1000 and 70 to 80 percent elimination of the organic compounds with a molecular weight >1000. More than 99 percent of the algae, expressed as chlorophyll, are eliminated during development periods. The decrease in turbidity is always higher than 99 percent. The very low remaining turbidity of <0.1 FTU, the small remaining concentrations of bacteria, algae, and iron (<30 µg/l) show that the quality of the filtrate is practically that of drinking water.

The decrease in P-input has caused a considerable decrease in the P-concentration in the Wahnbach Reservoir. As a consequence, there is less algal growth resulting in an increase in the Secchi depth as can be seen from Figure 5. The years 1969 and 1970 were chosen for comparison with the conditions after the commencement of phosphorus elimination as their hydrological conditions were similar. Not only Secchi depth increased, but there was also a considerable change in the composition of algal flora (Figure 6). Various species of tiny blue-green algae, e.g., *Coelosphaerium* and *Aphanothece*, completely disappeared after the plant went into operation. *Chlorella* which used to grow in large quantities is now present only in small amounts. In the spring diatoms such as *Asterionella formosa* and *Melosira* dominate.

ESTIMATION OF THE INFLUENCE OF P-ELIMINATION ON THE TROPHIC GRADE OF WAHNBACH RESERVOIR

If one tries to classify the Wahnbach Reservoir using the data from the OECD-Cooperative Program for Monitoring of Inland Waters (Vollenweider and Kerekes, in prep.),this reservoir would be with a probability of more than 50 percent mesotrophic in 1979 (Tables 2 and 3). During years of extensive *Oscillatoria* growth (e.g., 1969) with annual average chlorophyll concentrations of 25 µg/l, the reservoir was clearly eutrophic.

If one uses the total phosphorus concentration (annual average figures) instead of the average chlorophyll concentration, then the Wahnbach Reservoir would be with a probability of more than 50 percent oligotrophic.

If one applies the registered Secchi depths for classifying the reservoir, then the reservoir would be classed as oligotrophic or mesotrophic. The reservoir should be mesotrophic to eutrophic during the years 1969 and 1970 (Figure 5).

One should not forget that the data in Table 2 is based on the statistical evaluation of a large amount of data (Vollenweider and Kerekes, in prep.; Vollenweider, 1979). It is worth noting that the OECD cooperative program showed, for example, that the chlorophyll concentrations that actually occurred fluctuate to a far greater extent than the annual average values. This means that far higher concentrations of chlorophyll can occur for short periods of time in a mesotrophic lake. In 1979 peak concentrations of chlorophyll in the Wahnbach Reservoir were, however, only 10 µg/l.

Table 1. — Elimination of several substances by the phosphorus elimination plant (1.10.1977-31.5.1980).

Parameter	N	PEP-Inflow min.—max. x ± s	N	PEP-Outflow min.—max. x ± s	Elimination %
Coliforms bacteria/100 ml	560	0 – 68,000 5,979 ± 8,818	479	0 – 171 8 ± 15	99.87
Colony-count (22°C) colonies/ml	560	285 – 290,000 12,504 ± 20,530	479	0 – 17,100 263 ± 1,338	97.90
Chlorophyll µg/l	433	1.0 – 204.3 25.15 ± 27.69	360	0.1 – 17.3 1.28 ± 1.81	94.9
Turbidity FTU	515	0.6 – 48.7 10.4 ± 5.25	515	0.01 – 0.8 0.06 ± 0.09	99.3
COD mg/l	107	3.7 – 22.3 11.13 ± 4.52	97	0.1 – 6.3 2.56 ± 1.12	77.0
Spectral absorption coefficient 254 nm m^{-1}	561	3.4 – 20.8 8.14 ± 2.86	482	0.3 – 4.7 2.40 ± 0.68	70.5
DOC mg/l	563	0.9 – 7.3 2.37 ± 0.83	484	0.4 – 2.2 1.00 ± 0.30	57.8
Total P mg/m³	569	27 – 480 116.5 ± 49.2	485	1 – 13 4.3 ± 1.7	96.3

Table 2. — Preliminary classification of trophic state (OECD-Cooperative Program). The geometric mean (based on log 10 transformation) was calculated after removing values <or> x 2 SD.

		Oligotrophic	Mesotrophic	Eutrophic
Total Phosphorus mg/m³	x	8.0	26.7	84.4
	x ± 1 SD	4.85 - 13.3	14.5 - 49.0	48 - 189
	x ± 2 SD	2.9 - 22.1	7.9 - 90.8	16.8 - 424
	Range	3.0 - 17.7	10.9 - 95.6	16.2 - 386
Chlorophyll *a* (annual mean values) mg/m³	x	1.7	4.7	14.3
	x ± 1 SD	0.8 - 3.4	3.0 - 7.4	6.7 - 31
	x ± 2 SD	0.4 - 7.1	1.9 - 11.6	3.1 - 66
	Range	0.3 - 4.5	3.0 - 11.0	2.7 - 78
Transparency Secchi depth m	x	9.9	4.2	2.45
	x ± 1 SD	5.9 - 16.5	2.4 - 7.4	1.5 - 4.0
	x ± 2 SD	3.6 - 27.5	1.4 - 13.0	0.9 - 6.7
	Range	5.4 - 28.3	1.5 - 8.1	0.8 - 7.0

x geometric mean SD standard deviation (shortened from (10, 11))

SUMMARY

The phosphorus elimination plant developed by the Wahnbach Reservoir Association has been in operation at the point where the River Wahnbach flows into the reservoir since the end of 1977. Its operation has produced very clear water in the main reservoir and an average total phosphorus concentration of below 10 µg/l. This value was only exceeded for short periods of time, particularly when the inflow was higher than the retention capacity of the pre-reservoir and the efficiency of the plant.

The decrease in total phosphorus resulted in a shift in the algal species from blue-green algae to green algae and then to diatoms. This shift in species was typical of the change in the trophic state from eutrophic to oligotrophic-mesotrophic. Whereas blue-green algae have disappeared almost completely from the reservoir, the population of green algae has been reduced to such an extent that they have no dominating influence compared with diatoms. The clear decrease in the phosphorus input into the reservoir has caused a change in the trophic state of the lake which was eutrophic.

Table 3. — Classification of trophic state of the Wahnbach Reservoir (annual mean values (x)).

	Oligotrophic	Mesotrophic	Eutrophic
Total Phosphorus mg/m^3			
1969 without PEP		25	
1970		26	
1977		16	
1978 PEP in operation	9		
1979	6		
Chlorophyll *a* mg/m^3			
1969			25
1970			11
1977		7	
1978		8	
1979		5	
Transparency m Secchi depth			
1969			3
1970			3
1977	5		
1978	6		
1979	6		

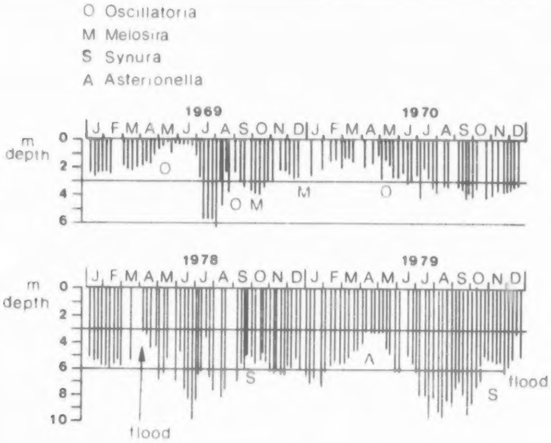

Figure 5. — Secchi-depths in the Wahnbach Reservoir before (1969/70) and after (1978/79) the begin of operation of the plant.

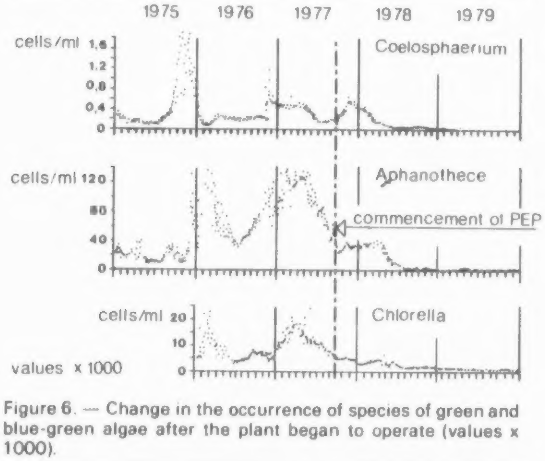

Figure 6. — Change in the occurrence of species of green and blue-green algae after the plant began to operate (values x 1000).

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AGRICULTURAL POLLUTION CONTROL IN THE NETHERLANDS

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ABSTRACT

The work of a Dutch Royal Commission to prepare "an inventory of agricultural pollutants disposed of - purposefully or inadvertently - into aquatic ecosystems" has taken some 8 years. I chaired a working group to quantify the disposal of chemical pollutants with manure and artificial fertilizers. The major efforts were directed to nitrogen- and phosphate pollution. In the study the peculiar structure of Dutch agricultural land had to be taken into account. The largest part lies below sea level with groundwater tables often 10 to 30 cm below soil surface; for agricultural use the groundwater table must be maintained at a fixed level. This means an export of rain water by pumping during winter and inlet of riverwater - mainly Rhine water - during summer. Thus the phosphate of the Rhine accounts for 50 percent of the input in the phosphate balance of Dutch waters. Other sources of phosphate come from layers of peat. Therefore, no reliable estimate could be made of the agricultural contribution because there are no areas where the (semi)natural input can be measured or quantified. However, partly due to the phosphate holding capacity of the soils the impression was obtained that neither manure nor artificial fertilizers contribute significantly to the phosphate input. This situation is completely different in the higher sandy soils, where intensive husbandry of cattle is performed. Considerable quantities of phosphates enter the waters in these regions. The situation is again different for the nitrogen balance. Considerable quantities of nitrogen reach the canals, lakes, and rivers, both in the form of ammonia and nitrate. Quantitave assessment of these data was not possible. The same difficulties as for the phosphate studies were met, while possible denitrification in the soil appeared to be an unknown factor of some importance. No proposals have been formulated for a control of these inputs. A manure balance for the whole country was established; no excess of manure seems to exist.

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URBAN STORMWATER/COMBINED SEWAGE MANAGEMENT AND POLLUTION ABATEMENT ALTERNATIVES

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ABSTRACT

Overflow points are the built-in inefficiencies of combined sewers. Untreated overflows from combined sewers are a serious water pollution source during both wet and dry weather periods. In urban areas, the principal nonpoint source concern is stormwater management. A nationwide survey of public works officials in 1976 identified urban flooding and its associated pollutants caused by inadequate storm sewers as the number one urban problem. As suburban land continues to be developed, the problems with separate storm and combined sewer systems will increase. Although stormwater runoff and combined sewer overflows typically occur only during brief periods, the quantities of sediment, nutrients, chemicals, and toxic metals dumped into streams during the storm periods dwarf the quantities of such materials released by the municipal treatment plants throughout the entire year. This problem has serious implications for communities using the streams for water supply as well as for other downstream users. Urban runoff management is a continuous process. Essential to its success is a constant process of innovation, demonstration, assessment, implementation guidance, and active program feedback. This paper reviews the innovative technology available today for implementation in our Nation's fight to protect and preserve our recreational receiving waters.

THE PROBLEM

Overflow points are the built-in inefficiencies of combined sewers. Untreated overflows from combined sewers are a serious substantial water pollution source during both wet and dry weather periods. Nationwide, there are roughly 15,000 to 18,000 combined sewer overflow points.

In urban areas, the principal nonpoint source concern is stormwater management. A nationwide survey of public works officials conducted in 1976 identified urban flooding and its associated pollutants caused by inadequate storm sewers as the number one urban problem. As suburban land continues to be developed, the problem will increase. In response, a few urban areas have initiated programs to improve stormwater management.

Although combined sewer overflows and stormwater runoff typically occur only during brief periods, the quantities of sediment, nutrients, chemicals, and toxic metals dumped into streams during the storm periods dwarf the quantities of such materials released by the municipal treatment plants throughout the entire year. This problem has serious implications for communities using the streams for water supply as well as for other downstream users.

Urban stormwater management is a continuous process. Essential to its success is a constant process of innovation, demonstration, assessment, implementation guidance, and active program feedback. Based upon January 1978 dollars, the total national needs to control pollution from combined sewer overflows were approximately $21.16 billion (U.S. EPA, 1978). Such a control program must be founded on proven capabilities, comparable methodologies and assessment criteria, an expanding data base, and a continuous, effective technology transfer.

Because of the unique nature of urban runoff abatement technology, control and/or treatment of storm sewer discharges and combined sewer overflows is a major problem in water quality management. Over the past 14 years much research has generated a large amount of data, primarily through the actions and support of the EPA's Storm and Combined Sewer Section.

Every metropolitan area of the United States has a stormwater problem, whether served by a combined sewer system (approximately 29 percent of the total sewered population) or a separate sewer system (Lager and Smith, 1974).

The problem is best quantified when discharges are compared on the basis of mass loadings released over discrete periods of time encompassing one or several consecutive storm events. In many cases, however, aesthetics or beneficial uses (such as maintaining receiving water quality above body contact use standards) are of primary concern.

Each metropolitan area should, therefore, be directly involved in setting its goals for a stormwater management program.

URBAN RUNOFF CHARACTERIZATION

Figure 1 illustrates representative strengths of wastewaters. The average 5-day biochemical oxygen demand (BOD$_5$) concentration in combined (domestic and storm) sewer overflow is approximately one-half the raw sanitary sewage BOD$_5$. However, storm discharges must be considered in terms of their shockloading effect. A common rainfall can produce flow rates up to 100 times dry-weather flow. Even separate stormwater is a significant source of pollution, having solids concentrations equal to or greater than untreated sanitary wastewater, and BOD$_5$'s approximately equal to secondary effluent. Bacterial contamination of separate stormwater is two to four orders greater than concentrations considered safe for water contact (Field, Tafuri, and Masters, 1977).

Because flow quantities are high, control — whether through flow balancing, multiple uses of facilities, runoff retardation, or combinations thereof — is the focus of cost-effective planning.

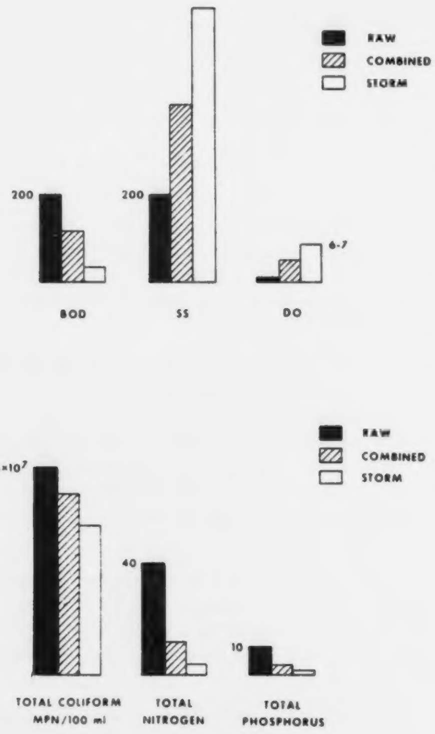

Figure 1. — Representative strengths of wastewaters (flow weighted means in mg/l).

THE ATTACK

The existing tools for reducing urban runoff pollution provide many-faceted approach techniques to individual situations. These tools are constantly being increased in number and improved upon as part of a continuing research and development program guided by the EPA Storm and Combined Sewer Section.

Continuing progress is being made in the variations and refinements of storage concepts. From the sophisticated computer controlled systems utilizing in-line storage capacities as found in Seattle and Detroit; monumental undertakings as the Chicago Tunnel and Reservoir Plan; Chippewa Falls off-line storage basin; Akron's underground void space storage; Sandusky, Ohio's underwater storage bag; and most recently, the evaluation of using static flow energy dissipators coupled with small off-line storage tanks and bulk-headed interceptors.

Simplified mathematical models based upon the general storage equation and operated off real (continuous) rainfall data provide an excellent tool for equating the effectiveness of alternate storage volumes and treatment rates.

Control and treatment of stormwater introduce many unique operation and maintenance requirements. These include automated control, startup and shutdown procedures, maintenance and surveillance between storms, and solids handling and disposal.

Much emphasis is currently being placed on controlling stormwater pollution by attacking the problem at its source, as opposed to potentially more costly downstream treatment facilities. These source controls, termed Best Management Practices (BMP), can either be directed toward planning control for further development or redevelopment efforts. The program has instituted research in using natural drainage features, erosion controls, operation and maintenance practices, highway deicing, street sweeping, collection system and catchbasin maintenance, and most recently, sewer flushing during dry weather to reduce receiving water impacts from first flush loads during storms.

Management alternatives for stormwater pollution abatement are generally categorized into four areas: Source control, collection system control, storage and treatment, and integrated (complex) systems.

SOURCE CONTROL

Source controls are defined as those measures for reducing stormwater pollution that involve actions within the urban drainage basin before runoff enters the sewer system. Examples include planning surface flow attenuation, using porous pavements, controlling erosion, restricting chemical use, and improving sanitation practices (street cleaning, more frequent refuse pickup, etc.).

Planning

Preventing and reducing the source of stormwater pollution best applies to developing urban areas, where man's encroachment is yet minimal, or at least controllable, and drainage essentially conforms to natural patterns. Such lands offer the greatest flexibility in preventing pollution. They must be developed in such a way that runoff remains close to natural levels. In these new areas proper management can prevent long-term problems.

On-site Storage of Runoff

The objective of on-site storage of runoff is either to prevent storm flow from reaching the drainage system or to change the timing of the runoff by controlling the release rate. Retention is the term for total containment, and detention is the common term for controlling the release rate to smooth out the peak flows.

The precipitation/infiltration process is the most important method of replenishing the groundwater reservoirs that serve as potable water supplies for many areas of the country. The decreased infiltration and increased water demand caused by urbanization will stress groundwater supplies unless recharge areas are set aside as basins develop. Although large-scale urban stormwater recharge programs have not been implemented because of potential groundwater pollution, on-site retention and recharge have been developed for small watersheds. Retention basins are usually variable-depth ponds designed with no outlet or only a bypass for exceptionally high flow conditions.

Retention is also practiced as controlled on-site storage where groundwater recharge is not important. In a typical example, the California Division of Highways has built retention basins to dispose of highway runoff in the San Joaquin Valley. These basins were developed from 0.4 to 2.4 hectare depressions that had originally been excavated for embankment material. Infiltration capacity is sometimes improved by excavating 1.8 to 3.1 meter deep trenches or vertical drains and backfilling with porous material. Maintenance is minimized by providing low-velocity channels ahead of the basins to help settle suspended particles. The areas are scarified once a year to decrease the surface clogging effects of organic solids.

A demonstration in Cleveland, Ohio (funded by Region V Great Lakes National Program Office with technical guidance from SCSS) is trying to obtain a quanitity and quality control on a portion of a combined sewer system by reducing overflows to receiving waters during rainfall events, and reducing residential basement flooding caused by combined sewer surcharging (U.S. EPA, 1970). Stormwater runoff is prevented from entering the already overburdened combined sewer as shown in Figure 2. Based upon applying the Dorsch HVM computer model to simulate runoff and backwater effects for each sub-catchment area, four upstream off-line storage tanks have been strategically placed to detain stormwater from entering the already surcharged combined sewer. The four units are constructed of corrugated structural steel pipe and measure: 2 tanks 156′ × 87″ × 63″; 163′ × 48″; 170′ × 93″ × 67″. Catchbasins are directing surface and street runoff into the tanks from which the stormwater will be discharged at a controlled rate into the combined sewer. The flow rate of the discharge will be regulated by a Hyrdrobrake internal energy dissipator. This small device located at the downstream end of the tank will deliver virtually a constant discharge rate regardless of head variations. This is accomplished without moving parts or external energy sources.

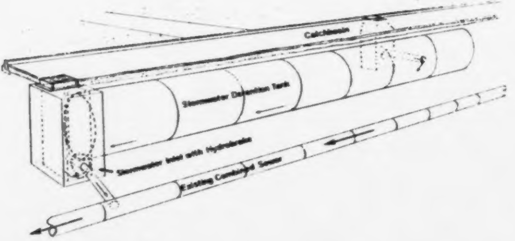
Catchbasin
Stormwater Detention Tank
Stormwater Inlet with Hydrobrake
Existing Combined Sewer

Figure 2. — Stormwater detention tank with hydrobrake.

Approximately 22 3 and 4 ″ diameter Hydrobrakes will be placed in existing catchbasins upstream from the larger detention tanks to maximize storage capacities and surface ponding.

Porous Pavement

An interesting technological answer to the problem of preserving pervious area is paving with an open graded asphaltic concrete. Experiments have shown that it will serve as a porous pavement, allowing as much as 64 cm/hr of stormwater to infiltrate through the pavement (see Figure 3).

Preliminary investigations have shown that this material can withstand stability, durability, and freeze-thaw tests, and that it compares in cost with conventional paving with drainage. Long-term tests will have to be made of its resistance to clogging and the effects on the quality of water that filters through the pavement. If the soil under the pavement and base is free draining, the rainwater will infiltrate quickly into the ground; however, porous pavement can also serve as a ponding device if storm quantities exceed soil capacity. The porus nature of the pavement permits water to be stored in the pavement. A pavement with a 10 cm surface course and 15 cm base course could store 6.1 cm of runoff in its voids (Thelan, et al. 1972). The proven use of porous pavement can be an important tool in preserving natural drainage.

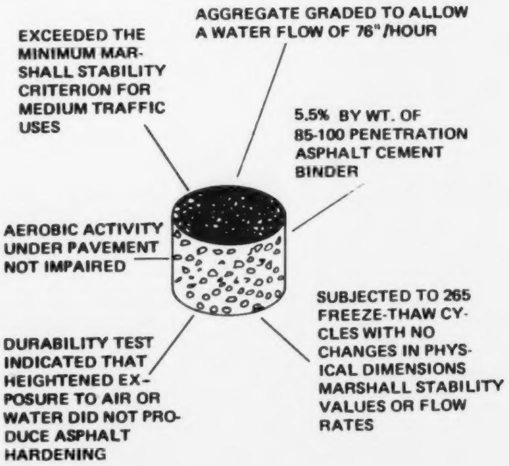
EXCEEDED THE MINIMUM MARSHALL STABILITY CRITERION FOR MEDIUM TRAFFIC USES
AGGREGATE GRADED TO ALLOW A WATER FLOW OF 76"/HOUR
5.5% BY WT. OF 85-100 PENETRATION ASPHALT CEMENT BINDER
AEROBIC ACTIVITY UNDER PAVEMENT NOT IMPAIRED
SUBJECTED TO 265 FREEZE-THAW CYCLES WITH NO CHANGES IN PHYSICAL DIMENSIONS MARSHALL STABILITY VALUES OR FLOW RATES
DURABILITY TEST INDICATED THAT HEIGHTENED EXPOSURE TO AIR OR WATER DID NOT PRODUCE ASPHALT HARDENING

Figure 3. — Porous asphaltic-concrete features.

Surface ponding is the most common form of detention being used by developers. In most cases, the facilities are carefully planned so that the ponding area is a dual-use facility that enhances the value of the site. Variable level ponds have a permanent water level during dry weather and increased holding capacity during storm conditions. The permanent lakes have aesthetic and recreational appeal, increasing lot values. Basins that are dry between storms are often designed to be used as baseball fields, tennis courts, and general open space. Parking lots can serve as low-depth storage ponds by sloping the sides and constructing drain outlets. Side slopes are restricted to about 4 percent for traction in the winter, and the pond depth is limited by the need for people to reach their vehicles during storm events. Obviously, a truck terminal lot can be allowed to pond to a greater depth than a supermarket lot. Table 1 indicates various surface ponding locations.

EROSION CONTROLS

Controlling erosion from construction and developing sites will have a major impact on the total pollution loads imposed on receiving waters. Current estimates indicate that approximately 3,900 km^2 (1,500 mi^2) of the United States is urbanized annually. All of this land is exposed to accelerated erosion (White and Franks, 1978).

From a knowledge of erosion and the guidelines that have been written concerning erosion control, several basic principles for control of erosion are apparent:

1. Reduce the area and duration of soil exposure.
2. Protect the soil with mulch and vegetative cover.
3. Reduce the rate and volume of runoff by increasing infiltration rates and surface storage and by diverting excess runoff.
4. Diminish runoff velocity with planned engineering works.
5. Protect and modify drainage ways to withstand concentrated runoff resulting from paved areas.
6. Trap as much sediment as possible in temporary or permanent sedimentation basins.
7. Maintain completed works and assure frequent inspection for maintenance needs.

These principles can be implemented by a variety of simply constructed facilities. Detailed descriptions and design criteria are available in the literature. Costs for some of the basic erosion control alternatives are presented in Table 2.

Chemical Use Control

One of the most often overlooked measures for reducing pollution from stormwater runoff is reducing the indiscriminate use and disposal of toxic substances such as fertilizers, pesticides, oil, gasoline, and detergents.

Table 2. — Erosion control costs per developed acre.

Vegetative measures	Initial placement cost, $/acre	First year maintenance cost, $/acre
Seeding: seedbed preparation, seed and application, mulching at 2 tons/acre		
Temporary seeding by machine	240-330	50-120
Temporary seeding by hand	335-415	50-120
Permanent seeding by machine	790-1,220	50-120
Sodding, including seedbed preparation	2,400-3,600	240-2,900
Mulch, 2 tons/acre		
By hand	120-140	---
By machine	90-120	---
Mechanical measures		
Earth diversion berms	0.15-0.30	1.20-3.60
Straw bale barriers	0.75-1.10	1.20-3.60
Silt basins with earth dam, watershed area		
2 acres to 5 acres	600-1,200	500-750
25 acres to 100 acres	1,200-3,500	750 -1,200
100 acres to 200 acres	3,500-5,000	1,200-1,800

$/acre x 2.469 = $/ha
acre x 0.405 = ha
tons/acre x 2240 = kg/ha

Table 1. — Surface ponding.

Site	Description	Cost estimate, $ With surface ponding	Cost estimate, $ Without surface ponding
Earth City, Missouri	A planned community including permanent recreational lakes with additional capacity for storm flow	2,000,000	5,000,000
Consolidated Freightways, St. Louis, Missouri	A trucking terminal using its parking lot to detain storm flows	115,000	150,000
Ft. Campbell, Kentucky	A military installation using ponds to decrease the required drainage pipe sizes	2,000,000	3,370,000
Indian Lakes Estates, Bloomington, Illinois	A residential development using ponds and an existing small diameter drain	200,000	600,000

Operations such as tree spraying, weed control, and fertilization of parks and parkways by municipal agencies, and the use of pesticides and fertilizers by individual homeowners can be controlled by increasing public awareness of the potential hazards to receiving waters, and providing instruction as to proper use and application. In many cases over-application is the major problem, where moderate use would achieve equal results. The use of less toxic formulations is another alternative to minimize potential pollution. Direct dumping of chemicals, crankcase oil, and debris into catchbasins, inlets, and sewers is a significant problem that may only be addressed through educational programs, ordinances, and enforcement (Am. Publ. Works Assoc. 1969).

Street Sweeping

Street sweeping is used by most cities to remove accumulated dust, dirt, and litter from street surfaces, but cleaning is usually done for aesthetic reasons. In many neighborhoods the amount of paper tolerated by the public governs cleaning frequencies. Street cleaning practices have been shown to be an effective way to attack the source of stormwater-related pollution problems.

Removal rates as reported in the literature vary considerably. In one study, the range was from 11 to 62 percent of the initial solids loading (McGuen, 1975). In another study, overall removal has been estimated at 33 percent of all pollutants on the street surface (McPherson, 1976).

Litter Control

Discarded containers from food and drink, cigarettes, newspapers, sidewalk sweepings, lawn trimmings, and a multitude of other materials become street litter. Unless this material is prevented from reaching the street or is removed by street cleaning equipment, it often is found in stormwater discharges. Enforcement of antilitter laws, convenient location of sidewalk waste disposal containers, and public education programs are just some of the source control measures.

COLLECTION SYSTEM CONTROL

Collection system control includes all alternatives pertaining to collection system management. Examples include inflow/infiltration control, the use of improved regulator devices, temporarily increased line-carrying capacities using polymer (friction reducing) flow additives, catchbasin maintenance, sewer separation, the use of remote monitoring/control systems, and the flushing of combined sewers during dry-weather periods.

Detailed knowledge of how collection systems respond to wet-weather flow is almost universally lacking in municipalities today. As a result, demonstration projects frequently reveal previously unknown relief points and crossovers critical to proper functioning. Such conditions emphasize the need for early and intensive monitoring and modeling for predictive responses.

Inflow and Infiltration

Extraneous flows entering a sewer can be generally categorized as either inflow or infiltration. Inflow usually occurs from surface runoff via roof connections, cross connections between sanitary and storm sewers, yard drains, or flooding of manhole covers. Infiltration usually occurs by water seeping into the pipe or manholes from leaky joints, crushed or collapsed pipe segments, leaky lateral connections, or other pipe failures. By reducing effective collection system and treatment plant capacities, extraneous flow may cause unnecessary pollution (Sullivan, et al. 1977). Table 3 presents rehabilitation cost estimates.

Table 3. — Rehabilitation cost estimates for inflow elimination.

Inflow source	Flowrate, gal/min	Rehabilitation cost (ENR 2000), $
Leakage around manhole covers	10-20	50-75
Holes in manhole covers	50-100	100-125
Foundation drains	10	300-1200
Roof leaders	10	50-75
Cross connection	250-450	100-500
Catchbasin	300	3000-5000
Ditch or storm sewer-infiltration sanitary sewer (per manhole reach)	60-80	500-2500
Area drains	50-200	50-350

gal/min x 0.0631 = L/s

Stormwater Regulations

The *swirl regulator/concentrator* is of simple angular-shaped construction and requires no moving parts. An isometric view of the final form of the device is shown in Figure 4. Again, the swirl provides a dual function: regulating flow by a central circular weir spillway while simultaneously treating combined wastewater by swirl action, separating solids from liquid. Dry-weather flows are diverted through a cunette-like channel in the floor of the chamber into the bottom orifice or foul underflow (located near the water downshaft) to the intercepting sewer for subsequent treatment at the municipal plant. During higher flow storm conditions, the low-volume concentrate (3 to 10 percent total flow) is diverted via the same bottom orifice leading to the interceptor, and the excess, relatively clear, high-volume supernatant overflows the central circular weir into a downshaft for storage, treatment, or discharge to the stream. This device is capable of functioning efficiently over a wide range of combined sewer overflow rates and can separate settleable lightweight matter and floatable solids at a small fraction of the detention time normally required for sedimentation. Figure 5 shows estimated costs for the swirl and helical bend units (Masters and Field, 1977).

The *helical bend flow regulator* is based on the concept of using the secondary helical motion imparted to fluids at bends, employing a total angle of approximately 60 degrees and a radius of curvature equal to 16 times the inlet pipe diameter (Sullivan, et al. 1975).

Figure 6 illustrates the device. The basic structural features of the helical bend are: The transition section from the inlet to the expanded straight section before the bend; the overflow side weir and scum baffle (not shown); and the foul outlet for concentrated solids removal and controlling the amount of underflow going to the treatment works.

Dry-weather flow goes through the lower portion of the device and outlets to the intercepting sewer and onto treatment. As the liquid level increases from storm conditions, secondary helical motion begins and the polluted solids are drawn to the inner wall and drop to the lower level of the channel leading to the treatment plant. As with the swirl, the proportion of concentrated discharge will depend on the particular design. The relatively clean combined sewer overflow passes over a side weir and discharges to the receiving waters, storage and/or subsequent treatment. Floatables are prevented from overflowing by a scum baffle along the side weir; they collect at the end of the chamber and are conveyed to the treatment plant when the storm flow and liquid level subside.

The hydraulic model studies of the helical bend regulator indicate that this flash method of solids removal can efficiently remove settleable solids with reasonably sized units and without using mechanical appurtances. Although its costs are greater than the swirl, structural and hydraulic head requirements may render it more appropriate.

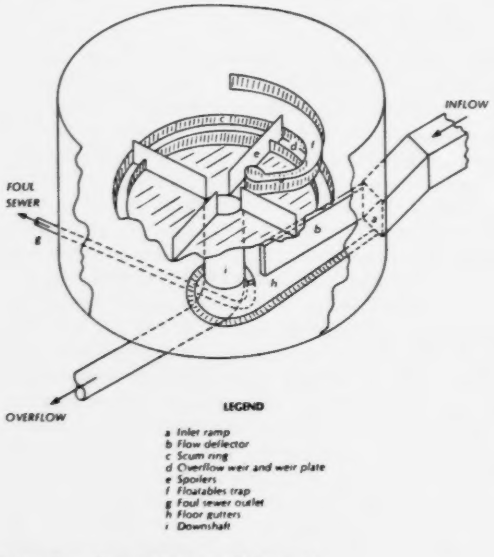
INFLOW
FOUL SEWER
g
OVERFLOW
a
b
c
d
e
f
h
i
LEGEND
a Inlet ramp
b Flow deflector
c Scum ring
d Overflow weir and weir plate
e Spoilers
f Floatables trap
g Foul sewer outlet
h Floor gutters
i Downshaft

Figure 4. — Isometric view of the swirl.

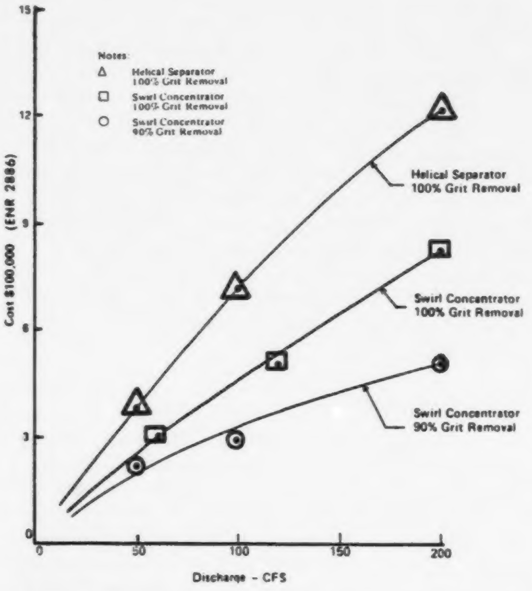

Figure 5. — Estimated construction costs — helical bend and swirl concentrator regulator.

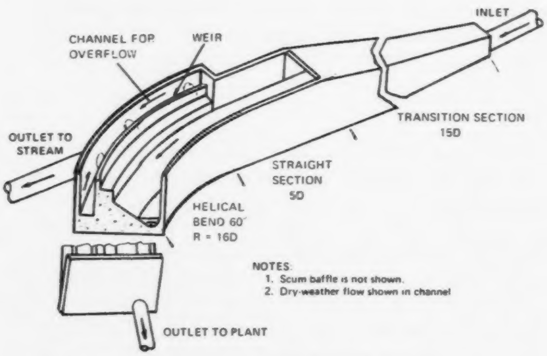
INLET
CHANNEL FOR OVERFLOW
WEIR
OUTLET TO STREAM
TRANSITION SECTION 15D
STRAIGHT SECTION 5D
HELICAL BEND 60° R = 16D
NOTES:
1. Scum baffle is not shown.
2. Dry-weather flow shown in channel
OUTLET TO PLANT

Figure 6. — Isometric view of helical bend regulator.

Catchbasin Maintenance

A catchbasin is defined as a chamber or well, usually built at the curbline of a street, for admitting surface water to a sewer or subdrain; at its base is a sediment sump designed to retain grit and detritus below the point of overflow. The distinction is made between catchbasins as devices which intentionally trap sediment and storm inlets which do not have sumps and as a result should not retain sediment.

Historically, the role of catchbasins was to minimize sewer clogging by trapping coarse debris and to reduce odor emanations from low-velocity sewers by providing a water seal. With improvements in steet surfacing and design for self-cleaning velocity in sewers, their benefits were considered marginal as far back as 1900. Despite the purported reduced need, catchbasins are still widely used. Catchbasins receive pollutants through the washoff of street surfaces and deliberate dumpings of crankcase drainings, leaves, grass clippings, pet feces, etc. (Lager, et al. 1977a)

Cleaning methods fall into four main categories: Hand cleaning, bucket cleaing, educator cleaning, and vacuum cleaning. Comparison of American Public Works Association survey data from 1959 to 1973 show that, on a national basis, the median cleaning frequency has decreased from twice per year to once per year. This trend is obviously detrimental from a water quality aspect; many problems associated with catchbasins may be traced to inadequate maintenance.

In general, catchbasins should be used only where there is a solids transporting deficiency in the downstream collection drains or at specific sites where surface solids are unusually abundant (such as beach areas, construction sites, unstable embankments, etc.). The advantages to be considered in converting existing catchbasins to inlets are (1) a direct reduction in the "first flush" pollutant load, (2) a reduction in required maintenance, and (3) the opportunity to relocate the conserved labor. Where catchbasins are required, they should be cleaned more often than once a year to limit the sediment buildup to 40 to 50 percent of the sump capacity. Figure 7 shows average solids removal efficiencies for catchbasins and the recommended design configuration.

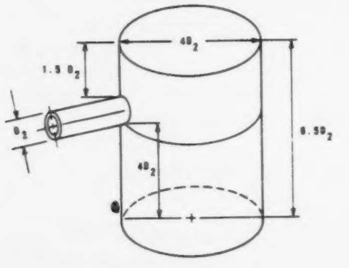

Figure Recommended design.

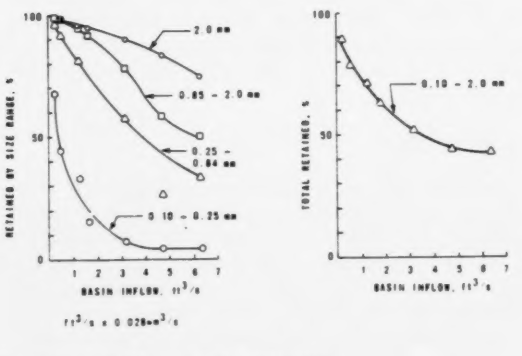

Figure 7. — Solids removal efficiencies.

SEWER FLUSHING

Regular flushing of sewers can ensure the continuing capability of sewer laterals and interceptors to carry their design capacity as well as alleviate the solids buildup that pushes solids into overflow. Sewer flushing can be particularly beneficial on sewers with very flat slopes (i.e., too flat for average flows to maintain sand and grit particles, with their associated contaminants in suspension at all times). If a small quantity of water is discharged through these flat sewers periodically, small accumulations of solids can be washed from the system. This cleaning technique is generally effective only on freshly deposited solids (Pisano, et al. 1979).

Internal automatic flushing devices have been developed for sewer systems. An inflatable bag is used to stop flow in upstream reaches until a volume capable of generating a flushing wave is accumulated. When the correct volume is reached, the bag is deflated by a vaccum pump releasing impounded water.

STORAGE

Storage facilities possess many attributes desired in stormwater treatment: (1) They may equalize flow and, in the case of tunnels, provide flow transmission; (2) they respond without difficulty to intermittent and random storm behavior; (3) they are relatively unaffected by flow and quality changes; and (4) frequently, they can be operated with regional dry-weather flow treatment plants for benefits during dry- and wet-weather conditions.

Storage facility variations include concrete holding tanks, open basins, tunnels, underground and underwater containers, underground "silos," granular packed beds (void space storage), abandoned facilities, and existing sewer lines.

System controls using in-line storage represent promising alternatives in areas where conduits are large, deep, and flat (i.e., backwater impoundments become feasible) and interceptor capacity is high. Reported costs for storage capacity gained in this manner range from 10 to 50 percent of the cost of similar off-line facilities. Because system controls are directed toward maximum utilization of existing facilities, they rank among the first alternatives to be considered.

Constructing new separate sanitary sewers to replace existing combined sewers largely has been abandoned because of the enormous cost, limited effectiveness, inconvenience to the public, and extended time required for implementation.

Costs associated with in-line storage systems are summarized in Table 4. Costs include regulator stations, central monitoring and control systems, and miscellaneous hardware (Lager, et al. 1977b).

Off-line storage is used to attenuate storm flow peaks, reduce storm overflows, and capture the first flush, or provide treatment in the form of sedimentation when storage capacity is exceeded. Off-line storage facilities may be located at overflow points or near dry-weather treatment facilities, depending on the type and function of the storage facility to be used. Off-line storage may also be used for on-site storage of runoff. Table 5 presents costs of off-line storage facilities (Lager, et al. 1977b).

Disadvantages of storage facilities include their large size, high cost, and dependency on other treatment facilities for processing the retained water and settled solids.

Table 4. — Summary of in-line storage costs[a].

Location	Storage capacity Mgal	Drainage area, acres	Capital cost, $	Storage cost, $/gal	Cost per acre, $acre	Annual operation and maintenance $/yr
Seattle, Washington						
Control and monitoring system	....	.....	3,500,000	....	...	73,000
Automated regulator stations	....	......	3,900,000	....	...	219,200
	17.8	13,120	7,400,000	0.42	564	292,000
Minneapolis-St. Paul Minnesota	NA	64,000	3,000,000	....	47	....
Detroit, Michigan	140	89,600	2,810,000	0.02	31	....

NA = not available
a. ENR 2000
$/acre x 2.47 = $/ha
$/gal x 0.264 = $/L
Mgal x 3785 = m^3

PHYSICAL TREATMENT ALTERNATIVES

Physical treatment alternatives are primarily applied to remove suspended solids from wastestreams, and are of particular importance to storm and combined sewer overflow treatment to remove settleable and suspended solids and floatable material. Physical treatment systems have demonstrated they can handle high and variable influent concentrations and flow rates and operate independently of other treatment facilities, with the exception of treating and disposing of the sludge/solids generated from these faciliites. The principal disadvantage is when equipment sits idle during dry weather. When implemented on a dual use basis as either pretreatment or effluent polishing of conventional sanitary sewage treatment plant flows, capital investment may be reduced by continuously using the physical treatment system.

Physical treatment processes that have been demonstrated on either a pilot or prototype scale include: Sedimentation and chemical clarification; solids concentration and flow regulation (swirl concentrator/flow regulator); screening; dissolved air flotation; high rate filtration; and a relatively new process, magnetic separation (Allen and Sargent, 1978). Many prototypes employ combinations of these processes to form integrated treatment systems, or use physical treatment processes in conjunction with biological and disinfection to produce desired water quality goals. Table 6 shows various removal efficiencies for physical treatment.

Biological Treatment

Biological treatment of wastewater, used primarily for domestic and industrial flows, produces an effluent of high quality at comparatively low cost. For treatment of storm flow, however, the following are serious drawbacks: (1) The biomass used to assimilate the waste constituents must either be kept alive during times of dry weather or allowed to develop for each storm event; and (2) once developed, the biomass is highly susceptible to washout by hydraulic surges and organic overload.

Examples of biological treatment applications to stormwater include (1) the contact stabilization modification of activated sludge, (2) high-rate trickling filtration, (3) bioadsorption using rotating biological contactors, and (4) oxidation lagoons of various types. The first three are operated conjunctively with dry-weather flow plants to supply the biomass, and the fourth approaches total storage of the flows (detention times of 1 to 10 days). Table 7 summarizes various biological treatment installations.

Integrated (Complex) Systems

The most promising approaches to urban storm flow management involve the integrated use of control and treatment systems with an areawide, multi-disciplinary (water use, land use, wet- and dry-period discharges, etc.) perspective.

Storm flow treatment processes can be most effectively used following some form of storage (flow equalization). This yields not only longer running periods, reduced shock effects, and buffer flexibility for startup and shutdown, but also, frequently, lower overall costs.

SUMMARY

Nonstructural and low structurally intensive alternatives offer considerable promise as the first line of action to control urban runoff pollution. By treating the problem at its source, or through appropriate legislation curtailing its opportunity to develop, multiple benefits can be derived. These include lower cost, earlier results, and an improved and cleaner neighborhood environment.

The greatest difficulty faced by BMP's is that the action-impact relationships are almost totally unquantified. It is clear that on-site storage, for example, can be closely related to reduced downstream conduit requirements but the net water quality benefits are far

Table 5. — Summary of off-line storage costs[a].

Location	Storage capacity Mgal	Drainage area, acres	Capital cost, $	Storage cost, $/gal	Cost per acre, $acre	Annual operation and maintenance $/yr
Akron, Ohio (21)	1.1	188.5	455,700	0.41	2,420	2,900
Milwaukee, Wisconsin (13)						
Humboldt Avenue	3.9	570	1,744,000	0.45	3,110	51,100
Boston, Massachusetts						
Cottage Farm Detention and Chlorination Station (17)[b]	1.3	15,600	6,495,000	5.00	416	80,000
Charles River Marginal Conduit Project (19)	1.2	3,000	9,488,000	7.91	3,160	97,600
New York City New York (22, 23, 25)						
Spring Creek Auxiliary Water Pollution Control Plant						
Storage	12.39	3,260	11,936,000	0.96	3,660	100,200
Sewer	13.00	.....	....	....	.....	....
	25.39	3,260	11,936,000	0.47	3,660	100,200
Chippewa Falls, Wisconsin (18)						
Storage	2.82	90	744,000	0.26	8,270	2,700
Treatment	.....	.....	189,000	.....	2,100	8,000
	2.82	90	933,000	0.26	10,370	10,700
Chicago, Illinois (2, 11, 26)						
Tunnels and pumping	2,998	240,000	870,000,000	0.29	3,630	......
Reservoirs	41,315	....	682,000,000	0.02	2,840	......
Total storage	44,313	240,000	1,552,000,000	0.04	6,470	......
Treatment	......	....	1,001,000,000	....	4,170	......
	44,313	240,000	2,553,000,000	0.04	10,640	8,700,000
Sandusky, Ohio (16)	0.36	14.86	520,000	1.44	35,000	6,200
Washington, D.C. (2, 15)	0.20	30.0	883,000	4.41	29,430	3,340
Columbus, Ohio (2, 3, 12)						
Whittier Street	3.75	29,250[c]	6,144,000	1.64	210	.....
Cambridge Maryland (14)	0.25	20	320,000	1.28	16,000	14,400

a. ENR 2000.
b. Estimated values; facilities under design and construction.
c. Estimated area.
$/acre x 2.47 = $/ha
$/gal x 0.264 = $/L
Mgal x 3785 = m^3

less defined. Similarly, cleaner streets and neighborhoods and enforced legislation will eradicate gross pollution sources but to what limit should these be applied and who will bear the cost? The final answers will not be found short of implementation.

However, one thing we can be assured of is that in view of the various documents which outline correct evaluation procedures and the continually developing state-of-the-art technologies, many local authorities will be able to significantly reduce urban runoff pollution in a cost-effective manner.

The technologies and procedures for combating stormwater pollution and combined sewer overflows are available today and are expanding rapidly. The EPA

Table 6. — Comparison of typical physical treatment removal efficiencies for selected pollutant parameters.

	Percent reduction					
Physical unit process	Suspended solids	BOD_5	COD	Settleable solids	Total phosphorus	Total Kjeldahl nitrogen
Sedimentation						
Without chemicals	20-60	30	34	30-90	20	38
Chemically assisted	68	68	45	......	..	..
Swirl concentrator/flow regulator	40-60	25-60	..	50-90	..	..
Screening						
Microscreens	50-95	10-50	35	......	20	30
Drum screen	30-55	10-40	25	60	10	17
Rotary screens	20-35	1-30	15	70-95	12	10
Disc strainers	10-45	5-20	15	......	..	..
Static screens	5-25	0-20	13	10-60	10	8
Dissolved air flotation[a]	45-85	30-80	55	93[b]	55	35
High rate filtration[c]	50-80	20-55	40	55-95	50	21
High gradient magnetic separation[d]	92-98	90-98	75	99	..	..

a. Process efficiencies include both prescreening and dissolved air flotation with chemical addition.
b. From pilot plant analysis.
c. Includes chemical addition.
d. From bench scale and small scale pilot plant operation, 1 to 4 L/min (0.26 to 1.06 gal/min).

Table 7. — Summary of typical biological stormwater treatment installations.

Project location	Type of biological treatment	Tributary area, acres	Design capacity, Mgal/d	Major process components	No. of units	Total size	Period of operation
Kenosha Wisconsin	Contact stabilization	1,200	20	Contact tank	2	32,700 ft³	1972 to 1975
				Stabilization tank	2	97,900 ft³	
Milwaukee Wisconsin	Rotating biological contactors	35	0.05[a]	3 ft diameter RBC units	24	28,300 ft²	1969 to 1970
Mt. Clemens, Michigan							
Demonstration system	Treatment lagoons in series with recirculation between storms	212	1.0[b]	Storage/aerated lagoon	1	750,000 ft³	1972 to 1975
				Oxidation lagoon	1	1,100,000 ft³	
				Aerated lagoon	1	930,000 ft³	
Citywide full-scale system	Storage/treatment lagoons in series with recirculation between storms	1,471	4.0[b]	Aerated storage basin	1	4,440,000 ft³	Under construction
				Aerated lagoon	1	508,000 ft³	
				Oxidation lagoon	1	1,100,000 ft³	
				Aerated/oxidation lagoon	1	922,000 ft³	
New Providence, New Jersey	Trickling filters	.....	6.0	High-rate plastic media	1	36 ft diameter	1970 to present
				High-rate rock media	1	65 ft diameter	
Shelbyville, Illinois	Treatment lagoons:						
	Southeast site	44	28[c]	Oxidation lagoon	1	255,600 ft³	1969 to present
	Southwest site	450	110	Detention lagoon plus 2-cell facultative lagoon	1	2,782,700 ft³	1969 to present
Springfield Illinois	Treatment lagoon	2,208	67	Storage/oxidation lagoon	1	5,330,000 ft³	1969 to present

a. Design based on average dry-weather flow; average wet-weather flow — 1 Mgal/d.
b. Design flowrate through lagoon systems. Total flowrate to facilities is 64 Mgal/d for the demonstration project and 260 Mgal/d for citywide system.
c. Estimated using a 50% runoff coefficient at a rainfall rate of 1.95 in/h.

acres x 0.405 = ha
Mgal/d x 0.0438 = m^3/s
ft^3 x 0.0283 = m^3
ft^2 x 0.0929 = m^2
ft x 0.305 = m
in/h x 2.54 = cm/h

recognizes the magnitude of the problem which is facing local communities and is ready to help in their fight to protect the quality of their receiving waters.

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THE GREAT LAKES: AN EXPERIMENT IN TECHNOLOGICAL INNOVATION AND INSTITUTIONAL COOPERATION

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ABSTRACT

Restoration and preservation of good water quality in the Great Lakes ecosystem is of great importance because of the magnitude of the resource and its many present and potential uses serving the basin's 40,000,000 population in the United States and Canada. Within U.S. EPA the Great Lakes National Program Office in Chicago serves as the coordinator and catalyst in dealing with Great Lakes problems, including the complex inter-media and inter-governmental aspects of this international resource. Congress has funded for staff and research under section 104(f) of the Clean Water Act and for demonstrations of pollution abatement technology under section 108(a). A variety of demonstrations is underway addressing agricultural nonpoint source control through improved techniques and management practices, and urban nonpoint sources through control of construction runoff, combined sewer overflows, and stormwater flows.

The Great Lakes National Program Office headquartered in the U.S. Environmental Protection Agency's Region V, Chicago, is responsible for managing and coordinating U.S. EPA's abatement and control programs as they affect the water quality of the Great Lakes. The Office serves as the Agency's catalyst to identify and recommend solutions to lakewide and transboundary pollution problems which cross cut traditional lines of authority. Externally, the Office is the principal U.S. focal point for communication, coordination, and cooperation for Great Lakes pollution issues with Canadian environmental agencies, the States, and the public.

The Office concentrates most of its scientific and technical resources on three key areas:

1. Revision and implementation of a Great Lakes monitoring program with particular emphasis on toxic organics, nutrients, and toxic metals.

2. Special investigations of serious "hot spot" problem areas, with emphasis on developing control measures for the full range of pollutant sources such as land, water, and air.

3. Increased State and public involvement in Great Lakes decisionmaking through the State/EPA agreement process.

Since the principal goal of U.S. EPA's Great Lakes effort is to restore and enhance water quality in the Great Lakes Basin ecosystem so that public health, welfare, and the environment are protected, the Great Lakes National Program Office relies heavily on the expertise of regional program offices, State pollution control agencies, and EPA research laboratories in finding solutions to these complex Great Lakes pollution problems.

It may be asked by those unfamiliar with either the breadth or the majesty of the Great Lakes ecosystem, why the special concern for these bodies of water? Size and use alone hold some of the answers. By volume, the Lakes contain 6 quadrillon gallons of fresh water — 20 percent of the world's fresh surface water and over 95 percent of the United States' supply. More than 40 million people — nearly 20 percent of the U.S. population and 50 percent of Canada's, live in the Great Lakes Basin. More than 23 million of these people depend on the Great Lakes for their drinking water. While those statistics in and of themselves indicate the vastness of the Lakes, they only begin to convey the problems which have resulted from such varied and intensive use.

The Great Lakes have been among the most abused waters in our country, and that abuse has had far-reaching effects. Since the area was first settled, the Great Lakes have been a convenient disposal site for every form of human waste and refuse. Industries, municipalities, and communities found it all too easy to discharge toxic substances, solid refuse and garbage, and biological wastes into the Great Lakes and the rivers feeding them. Runoffs from heavy rains and spring thaws of winter snows flowed into the streams, rivers, and the Great Lakes, carrying large amounts of fertilizers and pesticides with them. By the late 1960's, worldwide attention had focused on the severe contamination and pollution problems in the Great Lakes, which required direct and immediate action.

On the international scene, an institutional mechanism was already in place to guide those actions. Both Canada and the United States had long recognized the importance of the Great Lakes as a shared resource. In 1909, Canada and the United States signed the Boundary Waters Treaty, which concerns all the waters which form or cross the border between the two countries. The Treaty created the International Joint

Commission to deal with boundary water problems, including those of the Great Lakes. Many studies were conducted on the Lakes over the years and finally in 1972 the two Governments developed the first Great Lakes Water Quality Agreement. The IJC was asked to determine the pollution in Lakes Superior and Huron and also to determine the extent of pollution from land drainage. The studies were completed and reports submitted to the two Governments. In 1977-78 a review of the 1972 Water Quality Agreement was made and public hearings held to improve upon it.

In 1978 a new Water Quality Agreement was signed by the United States and Canada. The new agreement is more comprehensive than the first one in several ways. It includes the entire Great Lakes System — the land surrounding the Lakes, the streams flowing into them, and the Lakes themselves. It involves more than water quality. The ecosystem approach which recognizes the complex interrelationships among water, land, air and living things (plants, animals, and man) is found throughout the agreement.

The new agreement also emphasizes the need to understand and manage toxic substances. It reinforces the importance of controlling phosphorus pollution. It renews Government's commitment to control pollution from shipping and dredging, and to collect the data necessary to monitor water quality effectively. The Agreement of 1978 requires programs to determine the impacts and sources of airborne pollutants, and new measures to control pollution from various land uses. The agreement's general and specific objectives are designed to achieve and preserve a certain level of quality in the Great Lakes ecosystem.

But what have been some of the problems of the Great Lakes? In the 1960's enforcement conferences were held for each of the Great Lakes to determine their pollution status. Federal and State investigations involved water sampling and chemical/biological analysis to diagnose the Lakes' problems. It was determined then that nutrients were a major problem, especially in Lake Erie. Oil and grease, suspended solids, and organic contaminates were unsightly, damaging to wildlife, and caused problems with many water users. Untreated and/or inadequately treated industrial and municipal wastes were being discharged directly to rivers and lakes. Combined sewer overflows were causing bacterial pollution of beaches along with debris. Stormwater overflows were in some cases discharging toxic materials directly to surface waters. Schedules were set to remedy many of the problems but the law did not provide the teeth to enforce a cleanup effort.

In 1972 the Clean Water Act, Public Law 92-500, gave the U.S. EPA regulatory authority to enforce water pollution cleanup. Also during the 1970's other environmental laws were passed to further strengthen EPA's position. These laws included the Amendment to the Clean Water Act-1977, the Safe Drinking Water Act of 1974, the Resource Conservation and Recovery Act, the Toxic Substances Control Act of 1976, and the Clean Air Act Amendments of 1970 and 1977.

Back when Lake Erie was headlined as a "dead lake" and Rachel Carson's book entitled "Silent Spring" was stimulating environmental interest, State and Federal Governments concluded that phosphorus was the element that could best be controlled through waste treatment practices to reduce giant algal blooms in the lakes and the rapid aging taking place. Waste treatment processes were discussed and researched to see what could be done. Wastewater treatment requirements for municipal plants were set to provide secondary treatment with phosphorus removal. Industry was required to correct its discharge problems. In 1972 the Clean Water Act provided billions of dollars to upgrade municipal wastewater treatment plants to meet the Nation's pollution abatement needs.

Detergent phosphate bans were imposed in all of the Great Lake States but Ohio and Pennsylvania. Studies have indicated these bans significantly reduced phosphorus. At present a 1 mg/l effluent phosphorus limit is the target goal for wastewater treatment plants of 1 million gallons per day size or larger on the Great Lakes. To achieve the Agreement's target loadings may require not only greater point source control activity but also some nonpoint source controls.

Five billion dollars has been spent by EPA in the last decade to help clean up the Great Lakes. Additional billions of dollars have been spent by State and local governments and industries. While this expenditure of public and private funds has enabled us to abate the most visible Great Lakes pollution, it is what we do not see, taste, or smell that may cause severe problems in the years ahead. Clearly, the future challenges of lake restoration are in the area of toxic substance control.

The most serious threat is the existence of persistent toxic chemicals in Great Lakes' water, fish, wildlife, and sediments. These substances affect all portions of the Great Lakes in varying degrees. Many have the capacity to bioaccumulate; they have been found in the Lakes' fish and wildlife in alarming concentrations. Fish from Lake Ontario are heavily contaminated by Mirex. Lake Michigan fish cannot be sold commercially because of high levels of PCB's. Fish from Lake St. Clair had high levels of mercury that restricted their use for several years.

These substances reach the aquatic environment through direct discharges from industries, in runoff from agricultural and urban activities, and from the atmosphere after evaporation or insufficient incineration. While the effect of toxic substances on aquatic organisms is not well understood, severe adverse health effects on mammals and birds are well documented.

The National Program Office is checking the Lakes for toxic chemical "hot spots." One way we find these areas is through an extensive fish tissue and analysis program, which concentrates on fish found both in the open waters and in the nearshore tributary streams. Scientists combine findings from these surveys with results of intensive sediment studies to identify toxic chemical problem areas in selected harbors and tributary basins. We then use this information to identify specific sources and remedial measures. Regulation assessments are underway or planned in the following areas: the Ashtabula River in Ohio, Buffalo River in New York, Raisin River in Michigan, Indiana Harbor Canal in the vicinity of Gary, and Milwaukee, Wis.

But we are really only on the threshold of toxic substance control. We have some analytical and enforcement tools, but require much more. Our record is much better in the development of techniques to control conventional pollutants. We are striving to apply the knowledge gained in this area to the perplexing toxic questions. For example, the National Program Office administers the section 108(a) demonstration grant program which provides that the EPA can enter into agreements with any State, political subdivision, interstate agency, or other public agency to carry out projects to demonstrate new methods and techniques and to develop preliminary plans for the elimination or control of pollution, within all or any part of the watersheds of the Great Lakes drainage basins.

The Great Lakes National Program Office has entered into a number of demonstration projects with State and local entities of Government to develop and implement new methods and techniques for sediment and related pollutants from rural runoff and for reducing pollutants from urban runoff. Institutional and educational methods have been developed to help implement rural and urban nonpoint source pollution controls. Technical seminars have been held and project reports have been published for national distribution.

The section 108 program is closely coordinated with the section 208 water quality management planning and the Office of Research and Development. Data and technical information derived from Section 108 projects have impacted national nonpoint source guidance as well as States and local legislation and ordinances.

The section 108 program has been used to provide a systems approach to solving Great Lakes water pollution problems. We have tried to bridge the gap in EPA water pollution programs to tie planning and implementation together in one continuous effort. We use the planner, the institutional structure of State or local government, and citizen involvement. The program tries not to duplicate but rather enhance other EPA programs. An example would be Washington County, Wis., where the State Board of Soil and Water Conservation Districts was the grantee. They worked through the County Soil and Water Conservation District and the University of Wisconsin. Project Staff spent much time and effort with local public officials and land owners. Through these efforts and the use of grant funds to provide incentives for best managment practices demonstrations and to monitor results, individual involvement in nonpoint control efforts has been stimulated and some local governments have adopted construction runoff ordinances in the project area giving the Soil and Water Conservation District a role in reviewing subdivision plats.

Major water pollution problems in the Great Lakes that are high priority considerations for funding are as follows:

1. Toxic or hazardous substance control.
2. Combined sewer overflow pollution control.
3. Storm sewer overflow pollution control.
4. Rural nonpoint source pollution control.

To date this program has provided Congress and EPA Headquarters with data on nonpoint source pollution that have helped to develop the 1977 amendments to Public Law 92-500. Three major section 108 projects are the Black Creek project, the Washington County project, and the Red Clay project. Within these projects we have developed educational films and curricula for informing the public about nonpoint source pollution and the solutions to it.

We have developed and/or evaluated erosion control and a series of best management practices on the Black Creek Project that will improve water quality. We have also developed a watershed management model (ANSWERS) that accurately predicts sediment runoff during storm events; the model relates to the land management practices used, soil type characteristics, and slope of land.

The Black Creek Project began by evaluating 33 practices found in the Soil Conservation Service technical manual. A small number of the practices was found to be of major importance in the study area. Some sources originally thought to be important such as stream bank erosion turned out to be far less significant than others. Tillage practices, increasing crop residue and surface roughness, grassed waterways, livestock exclusion from streams, pasture planting, sediment control basins and terraces all proved to be of considerable use. A further general discovery was the importance of targeting critical areas rather than the original attempt of treating all areas.

The Washington County project investigators have provided much of the basic material and support that helped the State of Wisconsin pass its recent Sediment and Erosion Control legislation (Wisconsin Fund). All projects have achieved pollution reductions. Data and information from these section 108 projects have also been used in preparing the IJC Pollution from Land Use Activities Reference Group report and its remedial program recommendations.

Numerous technical reports have been published and distributed on section 108(a) activities. We have also encouraged some changes in Soil Conservation Service's procedures in dealing with land management practices as they affect water quality.

The National Program Office has worked closely with the Office of Research and Development at Edison, N. J., to demonstrate new techniques to reduce and remove pollutants from combined sewer overflows. We have three active projects dealing with urban combined sewer treatment and control.

At Rochester, N. Y. the Rochester Pure Waters District studied its drainage systems and developed a combined sewer overflow abatement program that recommended implementation of a best management practices system coupled with construction grant programs.

The demonstration of best management practices is underway and is scheduled to be completed about December 1980. This project involves the implementation and evaluation of minimal structural and nonstructural techniques to control urban storm and combined sewer overflow discharges. This represents the first phase of the master plan developed under a previous section 108(a) grant for the Rochester Pure Waters District. The best management practice program in conjunction with facilities provided under concurrent abatement programs is projected to result in an 80 to 90 percent reduction in the combined sewer overflow pollutant load to the Genesee River and the Rochester embayment of Lake Ontario. This is expected to significantly improve the water quality of several Lake Ontario-Rochester beaches.

At Saginaw, Mich. we are working with the Saginaw Department of Public Utilities to demonstrate that a swirl concentrator and degritter are acceptable, cost-effective methods for controlling and treating stormwater overflow from a combined sewer system. A study of the alternative treatment and control techniques to solve the City of Saginaw combined sewer overflow problem using swirl concentrators over retention basins can save the city $19,000,000 in capital costs and $206,000/year in operation and maintenance costs. This project is in its preconstruction stage. It is due to be completed in March of 1982.

At Cleveland, Ohio we are working with the City Department of Public Utilities and the Northeast Ohio Regional Sewer District to demonstrate control of sewer overflows during rain events. The process to be demonstrated is a controlled discharge of stormwater runoff according to the designed capacity of the individual sewer line. The catch basins are disconnected from the existing sewer and hooked into a detention tank from which the storm water will be discharged at a controlled rate by gravity through an internal energy dissipator (Hydro-brake) which requires no sources of energy, and has no moving parts. Capital savings in the order of 50 percent, compared to any conventional alternative for rehabilitation of combined sewer systems, are indicated. Evaluation of this process will start soon.

We are trying to get innovative technology demonstrated at the size and level such that consultant engineering firms will begin to factor these methods into their alternative treatment and control costing requirements under the municipal facilities planning exercise. We still have many gaps to bridge to get new technology into the system. We hope our projects can assist in filling this need.

But what of the future? If pollution contaminates more groundwater sources, even more millions of people will look to the Great Lakes as a source of drinking water. The energy situation may require that we use the Great Lakes even more intensively for navigation, power production, and possibly natural gas, for which Canada already drills in the western end of Lake Erie. Recreation close to home will continue; popular resort areas already face overbuilding and resulting strains on water treatment systems. Other emerging problems, such as increased levels of sodium and chlorides, also may affect the ecological balance within the Lakes and their interconnected systems.

Finding solutions to these problems requires both interstate and international partnership, a highly dedicated scientific community, and heightened public awareness. The key role of that public cannot be underestimated — for without their support, both financially and philosophically — the efforts to understand and help Lake processes may well be for naught.

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DESIGN OF STORAGE/SEDIMENTATION FACILITIES TO CONTROL URBAN RUNOFF AND COMBINED SEWER OVERFLOWS

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ABSTRACT

Urban stormwater runoff and combined sewer overflows are potentially significant sources of water pollution. Storage/sedimentation facilities have been recognized, both in the United States and Europe, as cost-effective measures for stormwater treatment and control. This paper summarizes a manual currently being prepared for the U.S. Environmental Protection Agency, detailing procedures for planning and design of various storage/sedimentation techniques. Such techniques as upland attentuation, inline storage, and end-of-pipe storage and treatment are detailed. Pollutants and watershed characteristics of stormwater management are discussed, including the range of water quality expected in urban stormwater runoff and combined sewer overflow. Data for the specific study area must be used. Models to evaluate the runoff problem and select effective solutions are listed. European practice in stormwater storage and sedimentation is described. Current practice in the United States in storage/sedimentation is discussed based on the American Public Works Association survey of 1980 and several case histories by Metcalf & Eddy. Recommended design practice is specified. Water quality benefits of both urban stormwater and combined sewer overflow storage/sedimentation are discussed.

INTRODUCTION

As municipal wastewater treatment is upgraded in accordance with the Federal Clean Water Act, urban stormwater runoff and combined sewer overflows are emerging as significant sources of surface water pollution in the United States. A 1975 survey of 56 public agencies located throughout the United States revealed that ". . . control (of) stormwater pollution from sources other than erosion . . . " ranked second only to flood control as a stormwater management goal (Poertner, Draft). The 1978 Needs Survey prepared by the U.S. Environmental Protection Agency estimated that $87.4 billion is needed by the year 2000 to bring combined sewer overflows and urban stormwater runoff into compliance with the requirements and goals of the Federal Water Pollution Control Amendments of 1972 (U.S. EPA, 1979).

Urban runoff is not a new problem. Traditionally, the goal of stormwater control has been to reduce or eliminate flooding. Temporary storage of runoff, a widely used method of flood control, is gaining wider application in the United States as a means of reducing the pollutant load of stormwater runoff. The U.S. Environmental Protection Agency is preparing a Design Manual for Storage/Sedimentation and Combined Sewer Overflows. This paper summarizes the contents of that manual, which will be available early in 1981.

THE MANUAL'S PURPOSE

In recent years, EPA has been committed to identify pollution sources other than municipal wastewater discharges and to develop viable methods for their control. A large amount of information has been developed over the past decade on stormwater runoff, and particularly, combined sewer overflow characteristics, receiving water impacts, and treatment. The Design Manual is to summarize the existing information and detail step-by-step procedures for stormwater storage/sedimentation treatment facilities. The Manual's audience is not only the hydrologist and stormwater control engineer, but the local decision-maker and land development engineer, as well.

The Design Manual is organized into six chapters. Chapter 1 is an introduction and guide to its contents. The second chapter, written for the nontechnical decisionmaker, overviews urban runoff and combined sewer overflow as pollution sources, and describes how storage/sedimentation facilities can be used to reduce the pollutant load. Chapters 3 and 4 outline the basic operating principles of storage/sedimentation facilities, and detail the data needs and design procedures for five types of facilities. Chapter 5 describes, through examples, the application of storage/sedimentation facilities in an overall stormwater management system. An important part of

Chapter 5 is devoted to explaining how existing flood control storage facilities can be retrofitted to provide better pollution control. The final chapter of the Manual draws heavily on European practice to suggest ways in which regional design guidelines can be developed and applied to controlling stormwater pollution.

A STORMWATER OVERVIEW

It is important when selecting or designing a stormwater control system to understand the types of pollutants contained in urban runoff and combined sewer overflows, the characteristics of the watershed that may influence the quantity and quality of stormwater, and the possible impacts of the stormwater on the receiving water.

Table 1 compares the concentrations of pollutants most commonly found in stormwater runoff with concentrations of the same pollutants in receiving water and sanitary wastewater. However, a wide variety of other pollutants, particularly toxic substances, may also be present in stormwater.

The pollutant concentrations shown in Table 1 should be used to identify relative magnitudes only. Runoff and combined sewer overflows are highly variable in the concentrations of pollutants present, as well as in quantities and rates of flow. Among the characteristics that may influence runoff quantity and quality from a watershed are hydrology, land use, physical characteristics of the surface such as soil type and percentage of area covered by impervious structures, and the type and configuration of the stormwater drainage system. The importance of collecting data on the stormwater runoff characteristics and treatability specific to the area cannot be overemphasized.

The severity of surface discharge of wastewater depends on the natural self-purification mechanisms of the receiving water. The goal of wastewater control is to reduce the pollutant load so that it can be assimilated without impairing the receiving water. In the United States, the impact of an urban stormwater discharge on the assimilative capacity of the receiving water must also be evaluated in light of other point and nonpoint discharges.

STORAGE/SEDIMENTATION OPTIONS

Generally, urban runoff and combined sewer overflow pollution occurs during periods of peak rainfall and runoff when the infiltration capacity of the ground surface and the transport and/or treatment capacities of the drainage system are exceeded. Temporary storage of stormwater runoff can reduce the peak rates of flow so that the transport and treatment capacities are exceeded less often. When the storage capacity is exceeded, storage basins may be designed to provide sedimentation treatment for the excess flow. Storage/sedimentation facilities can be categorized by disposal method. Detention storage facilities are those in which the stored runoff excess is released to the sewers at a reduced rate when capacity is available. The captured flows are usually treated before discharge when the storage aim is pollution reduction. Retention storage facilities capture flows which then are allowed to evaporate or percolate to the ground water without release from the facility.

DETENTION STORAGE FACILITIES

Three types of detention storage facilities are covered in the Design Manual: (1) Upland attenuation facilities, such as rooftop, parking lot, and plaza storage; (2) inline storage facilities; and (3) detention storage/sedimentation basins. The first two types are usually designed principally for storage. In most cases, maximum use of available upland and inline storage is made in combination with some downstream control facility. Detention storage/sedimentation basins are generally placed downstream of the storm or combined sewer system to provide both storage and sedimentation control.

Detention storage/sedimentation basins may be operated in a variety of modes. Excess runoff or combined sewer overflows are routed to the basins until the basins are full. At this point, all flows may continue to be routed through the basins, subjecting them all to sedimentation treatment. If a significant first flush is exhibited, as in small catchments with combined sewers, flows greater than the basin storage capacity may be bypassed to the receiving water. In this

Table 1. — Comparison of stormwater discharges to other pollutant sources. (mg/l unless otherwise noted.)

	TSS	VSS	BOD	COD	Kjeldahl nitrogen	Total nitrogen	Total PO_4-P	OPO_4-P	Lead	Fecal coliforms[a]
Background levels	5-100	--	0.5-3	20	--	0.05-0.5[b]	0.01-0.2[c]	--	<0.1	--
Stormwater runoff	415	90	20	115	1.4	3.10	0.6	0.4	0.35	13,500
Combined sewer overflow	370	140	115	367	3.8	9.10	1.9	1.0	0.37	670,000
Sanitary wastewater	200	150	200	500	40	40	10	7	--	--

a. ORGANISMS 100/ ML.
B. NO_3 as N.
c. Total phosphorus as P.

way, the first flush is captured without risking resuspension by later flows. The sediment captured in the basins is usually returned to the sewers when capacity becomes available for later treatment before discharge. The contents also may be released directly to the receiving water slowly so as not to exceed assimilative capacity.

When designing detention storage/sedimentation facilities, many factors are taken into consideration. Important data needs include watershed characteristics and hydrology, runoff pollutant concentrations and treatability, sewer and treatment plant capacities, and identification of available sites. A design procedure might follow these steps:

1. Quantify expected stormwater flows and pollutant loadings. Very often, computer simulation based on collected data is necessary. It is important that the distribution of runoff and pollutants within storm events be assessed.
2. Identify waste load reductions required. To ensure that receiving waters are protected, water quality impacts must often be assessed. Once the problem pollutants and required removal efficiencies have been identified, a decision can be made to design either for complete capture or for overflow sedimentation.
3. Identify feasible basin sites.
4. Capture basin design. Capture basins generally are located on very small catchments where a first flush of pollutants is most pronounced. The most important considerations are the degree of first flush exhibited, sewer and treatment capacities, and the removal of captured runoff and, particularly, solids after the runoff rate subsides.
5. Sedimentation basin design. Sedimentation basins can be very effective in removing particulate and floatable materials from urban runoff. The removal efficiency is a function of particle size and density, surface overflow rate, and horizontal velocity. Other important considerations include elimination of short circuiting, weir and basin depth design to cut down scouring of settled solids, and captured material removed.

RETENTION FACILITY DESIGN

The design discusses design principles and procedures for two types of retention storage facilities. Percolation/retention ponds, also called dry ponds, are earthen basins in which runoff is stored and allowed to percolate, usually within a few days. In wet ponds, excess runoff is stored in a permanent pond by varying the water level.

Retention storage facilities are generally very effective in reducing the pollutant loads, both suspended solids and BOD_5, for the runoff captured. They also have the added advantage of providing groundwater recharge. Because retention facilities depend on percolation and evaporation for emptying, these facilities are usually very large and shallow ponds. During overflow conditions, the deposited solids may be resuspended and carried over the overflow weir. Retention facilities are therefore most effective when operated as capture basins for first flush containment, with a total bypass of excess flows.

The data needs for designing retention storage facilities include watershed characteristics and hydrology, identification of available sites, and site soil characteristics. General design procedures include:

1. Quantify expected stormwater flows and pollutant loadings. The occurrence interval of runoff events is an important consideration, as well as runoff volumes and pollutant content. Retention facilities must be sized to allow sufficient emptying between events.
2. Identify the waste load reduction required. An assessment of receiving water impacts may be necessary or the required waste load reduction may be determined by a regulatory agency.
3. Identify feasible sites. The large area requirement and need for suitable soils are often the factors limiting the use of retention ponds.
4. Investigate the most promising sites for suitability of soils. It is important to keep in mind that silt from the runoff will tend to seal the soil surface and that the ponds must be sized according to the frequency with which the pond bottom will be scarified or dredged.
5. Quantify expected evaporative losses. For wet ponds, evaporation may be a major factor in the hydraulic balance.
6. Size the basins based on a water balance of all the hydrologic factors.

REGIONAL STORMWATER CONTROL GUIDELINES

For many generations, Europeans have used storage/sedimentation to control pollution from combined sewer overflows. In many cases, the Europeans have developed simple and easy to follow guidelines for designing these facilities. This approach is made possible because the guidelines are applied to very limited areas, in which storm patterns, land use, pollutant washoff functions, and water quality impacts are sufficiently similar to allow generalization. This same approach is being developed in some areas of the United States, such as Montgomery County, Md., Fairfax County, Va., and Denver, Colo.

The final chapter of the Manual looks at this regional guideline approach to stormwater management. It covers European practice in Scotland, Switzerland (Kanton), and Germany (Bavaria). Each is presented on a case study basis, including an evaluation of its effectiveness by regulations and agencies.

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SWEDISH EXPERIENCE OF NUTRIENT REMOVAL FROM WASTEWATER

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ABSTRACT

Water quality preservation steps in Sweden have been focused on chemically treating wastewater for phosphorus removal. In early 1979 more than 750 chemical or biological-chemical wastewater treatment plants were operating, treating about 75 percent of the total amount of wastewater from urban areas. This paper describes removal efficiencies for different process combinations, process improvements, sludge disposal, and treatment costs. The decreasing pollution load has improved many Swedish waters. Examples are given from brackish and fresh waters. Correlations between phosphorus and transparency indicate that reducing phosphorus will not markedly decrease the chlorophyll *a* content and thereby increase transparency until the phosphorus concentration in lake water is depressed below 0.1 to 0.2 g/m^3.

INTRODUCTION

During the 1950's and 1960's several Swedish lakes and coastal waters became markedly eutrophicated. Increasing population, increasing numbers of water closets and the use of phosphorus-containing synthetic detergents rapidly increased the P-load during a short space of time. The effects of household detergents were intensively discussed during the 1960's. Some modifications of these products reduced the P originating from them in sewage to approximately 30 percent (Natl. Swed. Environ. Prot. Board, 1972).

Water quality preservation steps in Sweden have focused on total nutrient removal, on expanding the chemical treatment of wastewater for phosphorus removal. This paper summarizes the development of sewage treatment, describes process combinations and efficiencies, efforts to improve treatment methods, sludge handling, and also gives examples of lakes where recovery has been observed following nutrient removal.

EXPANSION OF WASTEWATER TREATMENT PLANTS

The expansion of wastewater treatment plants in Sweden from the mid-1950's to the mid-1970's is described by Ulmgren (1975). The large extension of chemical sewage treatment began in 1968, with the purpose of reducing the phosphorus content in wastewaters. In early 1979 more than 750 municipal wastewater treatment plants were operating with chemical or combined biological and chemical treatment (Table 1), corresponding to about 75 percent of the total amount of wastewater from urban areas.

Table 1. — The number of sewage treatment plants and processes used in densely populated areas in Sweden, January 1, 1979 (Natl. Swed. Environ. Prot. Board, 1979).

Type of sewage treatment	Number of plants	Number of persons served
No treatment		7,000
Sedimentation	156	181,000
Biological	380	1,398,000
Chemical	141	324,000
Biological + chemical	625	4,833,000
Complementary	18	107,000
	1,320	6,850,000

QUALITY OF INCOMING WASTEWATER

The amount of wastewater entering a Swedish treatment plant is about 400 liters per person per day, including water from smaller industries, etc. Water consumption in households is about 200 liters per person per day.

Ulmgren (1975) analyzed incoming wastewater at 50 wastewater treatment plants (Table 2) and found the main change during the first half of the 1970's was a more than 20 percent decrease in the phosphorus content.

Table 2. — Quality of incoming wastewater to 50 Swedish treatment plants, g/m^3 (Ulmgren, 1975).

Parameter	Average value	Median value	Standard deviation	Number of analyses
Organic matter, BOD$_7$	123	116	± 60	122
Organic matter, COD	226	259	±115	78
Suspended solids	122	103	± 66	122
Total Phosphorus	5.7	5.5	± 2.5	124
Total Nitrogen	26	24	± 9	89

MONITORING SEWAGE EFFLUENT QUALITY

An effluent control program for studying the efficiency of the Swedish wastewater treatment plants has been directed by the Environment Protection Board. In particular, chemical oxygen demand (COD) and total phosphorus are being analyzed. At treatment plants serving more than 2,000 people, the samples for COD and total phosphorus are preserved in weekly flow proportional samples. The frequency of sampling increases with plant size. At plants >20,000 people, flow proportional, continuous sampling is conducted. At several plants samples are being taken continuously and analyzed by the minitest method (Elfring, Forsberg, and Forsberg, 1975).

The municipalities forward the results to the local county administration. The National Swedish Environment Protection Board then summarizes and evaluates the results annually (Natl. Swed. Environ. Prot. Board, 1979).

COMBINATIONS OF PROCESSES AND EFFICIENCIES

As illustrated in Table 1, the main process consists of biological and chemical treatment. Where poor receiving conditions prevail in relation to the discharge, complementary treatment, mainly in the form of postfiltration, is prescribed.

In early 1979 chemical sewage treatment was employed according to processes and sizes of treatment plants listed in Table 3. Post-precipitation dominated the treatment. At that time complementary treatment was used at 18 sewage works, serving about 100,000 people. Most of the sewage from the Stockholm area was treated by pre- or simultaneous precipitation.

Earlier, aluminum sulfate was the dominant precipitant. Today iron salts are also frequently used. At about 50 smaller plants, lime is the precipitating agent.

Table 3. — Flocculation processes and size distribution of wastewater treatment plants, January 1, 1979 (Natl. Swed. Environ. Prot. Board, 1979).

Flocculation process	Number of treatment plants designed for pe <500	501-2000	2001-5000	5001-20,000	>20,000	Total
Direct precipitation	28	71	28	10	4	141
Pre-precipitation	-	1	1	2	10	14
Simultaneous precipitation	3	13	4	7	6	33
Post-precipitation	58	194	137	119	69	576

Phosphorus removal efficiencies for different process combinations have been discussed recently (Gronquist, et al. 1978; Hultman, 1978,1979). In spite of similar processes, precipitants, size of load, etc., the results from different plants vary widely. This illustrates that factors not normally monitored have a great influence on the treatment efficiencies. In cases where there are no significant process disturbances, the phosphorus concentrations listed in Table 4 refer to permanent full scale operation. Hultman (1978) pointed out that the data in this table are comparatively old. New evaluations, at present being compiled, will probably change the ranges given in Table 4. Normally loaded plants operating with lime precipitation, for instance, seem to be more efficient than indicated.

Table 4 — Phosphorus removal efficiences for different processes. Modified after Hultman (1978).

Effluent P-concentrations g/m^3	Processes
0.5-1.2	Post-precipitation, Al-sulphate, pH 6.5-7.2 Post-precipitation, lime
0.5-0.8	Pre-precipitation Simultaneous precipitation
0.2-0.4	Post-precipitation, Al-sulphate, pH 5.5-6.4 Post-precipitation, lime (low loaded) Post-precipitation, Fe^{3+} + sludge recirculation Pre-precipitation, + filtration
0.15-0.3	Post-precipitation, Al-sulphate, pH 5.5-6.4 + filtration Simultaneous precipitation +contact filtration Pre-precipitation + contact filtration

DISPOSAL OF MUNICIPAL SLUDGE

The growing demand for more advanced wastewater treatment has considerably increased the volume of municipal sludge. Since 1960 the amount has increased about threefold (Tullander, 1975).

Sludge can be disposed of either at sludge disposal sites as landfill, or for agricultural use in enriching soil.

Using sludge for agricultural food production poses a number of hygienic and environmental hazards. The Swedish National Board of Health and Welfare has investigated this problem and published instructions in 1973. Standards for evaluating the quality of sludge are given by Tullander (1975).

Special attention has been devoted to the content of heavy metals. At present it is not possible to make a definite evaluation of the biological effects of these metals. As a general rule, frequent and long-term use of sludge on any one field should be avoided. Similarly, sludge having excessive levels of heavy metals should be avoided in agriculture. From an environmental viewpoint, the maximum amount of sludge spread on individual fields should not exceed 5 tons of dry solids/ha during a 5-year period. Declaration of contents is recommended for sludge. This will simplify adherence to the standards and give the treatment plants people valuable information on the composition of the wastewater and also on the need for further improvements of the treatment processes.

COSTS OF SEWAGE TREATMENT

The costs of sewage treatment have been examined by Hultman (1978). His values for post-precipitation are reproduced in Table 5, showing that capital costs are somewhat higher than operating costs. Centralizing the wastewater treatment in bigger plants will decrease treatment costs.

The chemicals necessary for nutrient removal require about 25 percent of the operating costs for post-precipitation. The cost of chemical precipitation is 15 percent and of sludge conditioning 10 percent.

Table 5. — Approximate costs for sewage treatment, 1978 (Hultman, 1978).

Number of person Equivalents (p.e.)	Costs for post-precipitation plants (including sludge treatment)		Additional costs for deep-bed filtration	
	Capital costs*	Operating costs*	Capital costs*	Operating costs*
2,000	130	100	-	-
5,000	100	70	25	8
20,000	60	50	10	4
50,000	45	40	7	3

* in Swedish Crowns per capita per year.

Notes: 1 Swedish Crown = 0.23 $
In calculation of capital costs the annuity used is 10% and 13% for post-precipitation plants and deep-bed filters, respectively.

PROCESS IMPROVEMENTS

After the rapid expansion of advanced wastewater treatment, efforts are now concentrated on reducing the operating costs and promoting efficiency. Important work is being done within the Nordic Cooperative Organization for Applied Research (NORDFORSK). A project concerning management of municipal wastewater treatment plants has resulted in five reports dealing with flow equalization in sewer systems (Stahre, 1978), wastewater filtration (NORDFORSK, 1978), evaluation of continuously operating measuring instruments (Holmstrom, 1979a), guidelines for monitoring programs (Balmer, et al. 1979), and simultaneous precipitation (Gronqvist and Arvin, 1979).

Very promising results have been obtained with methods where phosphorus is chemically reduced before the final precipitation step. Simultaneous precipitation followed by contact filtration gave an average effluent concentration of 0.24 g phosphorus/m^3. The operational cost at this small plant (about 2,000 people) was reduced by about $8,000 per year (Holmstrom, 1979b). Recirculation of post-precipitated sludge to the activated sludge process has improved effluent quality at reduced cost. Examples from Uppsala and Eskilstuna have been reported (Hultman, 1979; Forsberg, 1977), where the effluent phosphorus was decreased to about 0.3 g/m^3. The recirculation makes it possible to decrease the precipitant dose. In 1977 this reduced the cost for the precipitant in Uppsala (200,000 people) by about $60,000 (Forsberg, 1977). Sludge with improved settling and dewatering properties is also often obtained as a result of this recirculation of chemical sludge.

Two-stage precipitation, i.e., simultaneous precipitation followed by post-precipitation, also gave concentrations of effluent phosphorus corresponding to 0.3 g/m^3.

Other methods tested are regulating the alkalinity of the wastewater to reduce the requirement of aluminum sulfate or lime in post-precipitation, and using automatic control to save chemicals and energy (Hultman, 1979).

DECREASING POLLUTION LOAD

The comprehensive development of municipal wastewater treatment has markedly reduced the pollution load on Sweden's water courses and coastal waters. The biological oxygen demand (BOD_7) load was about 80,000 tons/year around 1960; the phosphorus load above 7,000 tons/year at the end of 1960. At the end of 1970 these figures had been lowered to about 20,000 and 2,500 tons/year, respectively (Falkenmark, 1977). For the city of Uppsala the phosphorus load has been reduced to that observed about 50 years ago (Forsberg, 1979), a situation probably prevailing in many cities served by advanced wastewater treatment. It must also be mentioned that intensified anti-pollution efforts within the industry have markedly contributed toward reducing the total pollution load on Swedish water bodies (Falkenmark, 1977). Both municipal and industrial anti-pollution measures have been supported by State grants.

RECOVERY OF POLLUTED WATERS

The decreasing pollution load has improved many Swedish waters. Table 6 shows decreasing phosphorus concentrations and increasing transparency in brackish and fresh waters in the Stockholm areas and in two of the largest lakes in Sweden. Transparency here mainly reflects algal turbidity. Because of the improved conditions in Lake Malaren, open air bathing has once again become possible in the most central parts of Stockholm.

To study more in detail the effects of nutrient removal, a comprehensive program was started by the National Swedish Environment Protection Board in 1972 for analyzing the loadings on and the conditions in a number of different recipient lakes (Forsberg, Ryding, and Claesson, 1975). Results from some lakes showing both improvements and delayed recovery have been presented (Ryding and Forsberg, 1976; Forsberg, et al. 1978). Results from 22 lakes have been evaluated and briefly summarized in Table 7. The majority of these lakes have responded positively with lowered concentrations of total phosphorus and organic matter. Half showed lowered chlorophyll *a* values, but in only six lakes did transparency increase significantly. Nitrogen increased in 10 lakes. Increased nitrogen values have also been observed in other Swedish waters analyzed during the 1970's. The four lakes showing decreasing nitrogen content are lakes where sewage has been totally diverted.

Correlations between phosphorus and chlorophyll *a* and between chlorophyll *a* and transparency have been presented for these lakes (Forsberg and Ryding, 1979). In waters where transparency is principally influenced by algal turbidity a correlation can be expected between phosphorus and transparency, at least within the concentration range where phosphorus is the primary algal growth-limiting nutrient. Similar correlations have also been demonstrated (Lee, Rast, and Jones, 1978). Table 6 indicates close correlations between these parameters.

Table 6. — Total phosphorus and transparency in the Stockholm area (Riddarfjarden, Blockhusdden, Tralhavet, Cronholm and Bennerstedt, 1978, central part of Lake Malaren (S. Bjorkfjarden) and Lake Vattern, Ahl, pers. comm.

Water body	Period	Total-P, g/m^3	Transparency, m
Riddarfjarden	1968-70	0.072	2.1
	1971-73	0.040	3.1
	1974-76	0.032	4.5
Blockhusudden	1968-70	0.173	1.9
	1971-73	0.091	2.0
	1974-76	0.049	2.2
Tralhavet	1968-70	0.060	2.4
	1971-73	0.048	2.6
	1974-76	0.027	2.9
S. Bjorkfjarden	1965-69	0.035	3.0
	1974-78	0.020	4.3
L. Vattern	1968-70	0.010-0.015	7-8
	1978-80	0.007-0.008	10-12

Table 7. — Change in water quality observed in 22 lakes after nutrient removal.

	Number of lakes				
Changes in concentrating	Total nitrogen	Total phosphorus	Organic matter	Chloro-phyll a	Transpa-rency
Decreasing	4	14	15	11	3
No significant change	8	7	7	9	13
Increasing	10	1	0	2	6

In heavily polluted (hypertrophic) lakes a reduction of phosphorus will not markedly decrease the chlorophyll *a* concentration and thereby increase transparency until the phosphorus concentration is depressed below 0.1 to 0.2 g/m^3. This is illustrated in Figure 1, where seasonal averages of phosphorus are plotted against the corresponding values of transparency for 12 of the 22 lakes evaluated and listed in Table 7. Analyses showed that a comparatively large change in annual P-load must occur, a reduction by about 70 percent of the pre-diversion data, to achieve any significant improvement in clarity in Lake Boren and Lake Ekoln (Forsberg, et al. 1978). A reduction by 30 to 40 percent revealed less marked improvements.

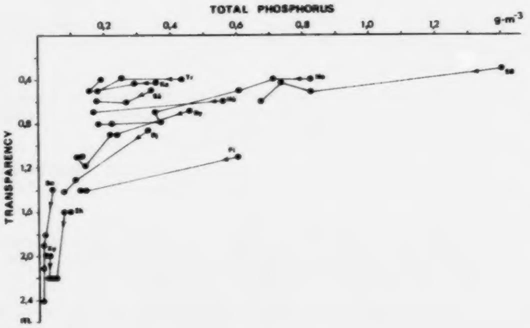

Figure 1. — Total phosphorus versus transparency in 12 wastewater receiving lakes. Surface water (0-2 m). Average values based on one sample/week, June-September. Bo = Lake Boren, Dj = L. Djulosjon, Ek = L. Ekoln, Fi = L. Finjasjon, Ha = L. Hacklsjon, Ka =L. Kalven, Ky = L. Kyrkviken, Ma = L. Malmsjon, Ry = L. Ryssbysjon, SB = L. Sodra Bergundasjon, Sa = L. Sabysjon, Tr = L. Trehorningen. For geographical positions see Forsberg and Ryding, 1979.

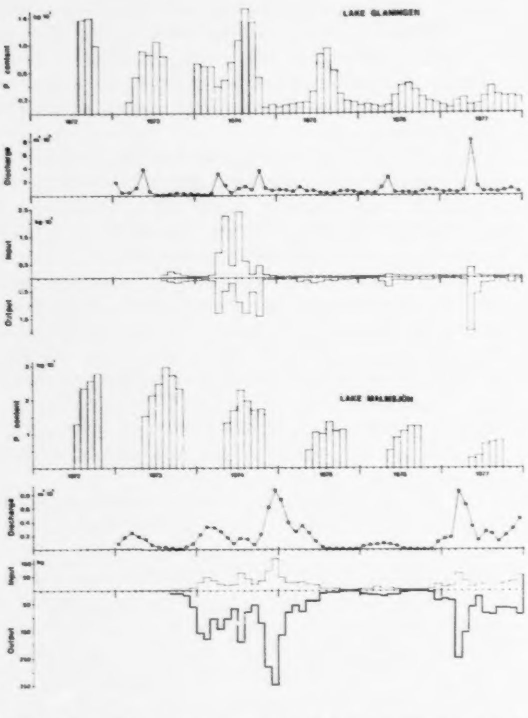

Figure 2. — Phosphorus content in lake water, discharge and external in- and output of phosphorus in Lake Glaningen and Lake Malmsjon after diversion of sewage in early 1974. Monthly average values, 1972-1977.

A great amount of phosphorus is known to be released — and recycled — from the sediments in shallow, polluted lakes. The recovery after decreased external nutrient load in these lakes will be delayed, as seen in several lakes treated in Table 7. A high water flow through a lake is favorable for washing out phosphorus, illustrated (Figure 2) by data from two lakes where sewage was totally diverted in 1974. Lake Glaningen, having a higher flushing rate than Lake Malmsjon, attained stable conditions more rapidly. The phosphorus decrease in Lake Malmsjon was still significant during the fourth summer period after the diversion. A similar trend has also been reported for Lake Norrviken (Ahlgren, 1977).

DISCUSSION

When the rapid development of advanced wastewater treatment for phosphorus removal started in Sweden about 10 years ago, the knowledge of biological-chemical treatment was comparatively limited. The rapid development was positive and valuable, greatly reducing pollution loads and improving the water quality in rivers, lakes, and coastal waters. Experiences obtained during this decade indicate, however, that several treatment plants do not operate as efficiently as expected. Depending on lack of operating experience and guidelines and unsuitable process technology and equipment, effluent values of total P were not below the required limit, 0.5 g/m^3 for about 40 percent of the plants (Hultman, 1978). Therefore, it seems necessary to improve or change processes to obtain better effluent quality at reduced operating cost.

As most of the Swedish population now is served by biological-chemical treatment plants, it seems unrealistic to expect that new advanced technologies will be introduced if they are not adaptable to already operating plants. Therefore, further efforts to improve the effluent quality and to reduce the operating costs will be concentrated on recirculation of chemical sludge, stepwise precipitation, and additional steps, e.g., by filtration. To maximize the efficiency it will also be important to have an effective emergency service in case of technical mishaps. It will be also necessary to have well prepared and competent employees handling the plants, especially when the treatment processes become more complex.

The recovery of polluted waters will be influenced by many different factors, such as hydrological and morphometrical conditions, the size and rate of nutrient reduction, the chemical and biological character of the recipient water, etc., making it a complicated process. For a more general evaluation of the effects of improved wastewater treatment, results and experiences over a long period and from a great number of different waters are needed.

The Swedish experiences show that natural waters can recover rapidly after nutrient removal, e.g., by advanced wastewater treatment (Forsberg, et al. 1978). To obtain visible results a comparatively large change in the annual P-load must occur. The improvement is easier to achieve in deep stratified waters (Table 6) than in shallow lakes influenced by internal loading from the sediments (Forsberg, 1979; Ahlgren, 1977; Bengtsson, et al. 1975; Ryding, 1978). In several shallow, polluted lakes the internal loading is a serious problem during the vegetation period, i.e., that period when public concern for good water quality is at a maximum (Ryding and Forsberg, 1977, 1980a; Forsberg and Ryding, 1980).

Strong winds induce an increased vertical mixing and thereby an increased internal loading, implying that climatic fluctuations have to be monitored when studying the response to nutrient removal measures in shallow lakes (Ryding and Forsberg, 1977; Forsberg, 1978). Multiplying the duration of critical wind directions by the force of the wind produces a close correlation between "stirring capacity" and the chlorophyll concentration (Ryding and Forsberg, 1980b).

During high phosphorus content in these shallow, internally loaded lakes (July-September) the water flow through the lake is often very low. To avoid recycling, a high flushing rate is therefore necessary for wash out of substantial amounts of phosphorus. To improve conditions in the shallow, hypertrophic Lake Finjasjon in Skane (south Sweden), where advanced wastewater treatment does not help, a temporary damming (within natural fluctuation levels) during July-August, followed by a rapid lowering of the water level has been suggested as a way to wash out comparatively large amounts of phosphorus (Ryding and Forsberg, 1980a).

Even in deeper and larger lakes the hydraulic residence time has been found to accurately describe the trophic state (Vollenweider, 1975; Sonzogni, Uttormark, and Lee, 1976). One way to further refine the nutrient load-lake response concept is to apply an estimate of the load that is more related to the actual hydrological conditions for each separate lake for the growing season compared to annual loading figures. Calculations of a so-called hydraulic relevant phosphorus load (i.e., the amount of imported phosphorus during the growth period and one "filling time" prior to it) adequately described the summer phosphorus content in Lake Boren (Ryding and Forsberg, 1980b). As is evident, Sweden has gained much experience in nutrient removal; results are both positive and negative. The big "cleaning up" occurred during a period of good economy. Today, costs and energy problems make it necessary to find new and more inexpensive methods. The Lake Finjasjon model (Ryding and Forsberg, 1980a) for lake restoration may be one approach.

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STORMWATER POLLUTION CONTROLS FOR LAKE MANAGEMENT

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ABSTRACT

This paper presents an overview of an on-going stormwater management project for Lake Quinsigamond, Mass. The overall project is funded by the National Urban Runoff Program (NURP) and Section 314 Lakes Restoration Program. Lake Quinsigamond is located in central Massachusetts and is the deepest manmade lake in the State. Presently, pollution point sources have been eliminated. However, urban/commercial growth is high in the watershed and eutrophication has been notably quickened by the accelerated stormwater solids, organic and nutrient loadings. It is envisioned that solids separation devices may be an important low-level structural control for eliminating high concentration urban runoff solids (and nutrient) loadings to the lake. Current settleability tests of collected stormwater samples are also described in this paper. These tests are meant to determine the relative fractions of easily removable floatable, settleable, and suspended solids versus the more difficult light, colloidal material.

BACKGROUND INFORMATION

Lake Quinsigamond is located in the middle of Massachusetts between the City of Worcester and the Town of Shrewsbury. The extreme southernmost portion of the lake lies within the Grafton boundary. Figure 1 shows Lake Quinsigamond and its tributary system. The lake lies in a north-south direction and is crossed by three major highways: Interstate 290, Route 9, and U.S. Route 20.

Lake Quinsigamond is separated into two distinct sections: The deep, narrow, northern basin and the shallow southern basin known as Flint Pond. The total area of the lake is 312 hectares comprised of 192 hectares in the northern basin and 120 in Flint Pond. The lake has a maximum depth of 26 meters and an average depth of 6 meters. The lake is approximately 8 kilometers long with the width varying from 76 meters to 1.6 kilometers. The lake volume is estimated at 19.43 million cubic meters.

Being situated in a highly urban area, the lake supports multiple recreational uses including fishing, boating, water skiing, and bathing. The entire periphery of the lake is densely settled with many private homes and some commercial establishments. Two State parks, several private beaches, and marinas are located along the shorefront.

Lake Quinsigamond presently meets the water quality standards required for water contact recreation. The main body of Lake Quinsigamond has passed through the mesotrophic stage and is in an early eutrophic stage. Although the water quality of the lake is satisfactory, intensive development of the drainage basin has accelerated the lake's natural aging process, and may limit the lake's recreational value in the future.

Stormwater runoff from the drainage basin is believed to be the major factor causing the accelerated rate of eutrophication. Stormwater contributes significant loadings of phosphorus and inorganic nitrogen to the lake. Stormwater carries large amounts of solids into the lake, increasing the turbidity of the lake water and creating sandbars that make boating hazardous and provide areas for rooted aquatic plant growth. Stormwater degrades the bacteriological quality of the lake.

Concern about the deteriorating water quality combined with the tremendous desire to use the recreational assets of the lake has produced widespread concern for the future of Lake Quinsigamond. Consequently, over the last several years investigations of the water quality to the lake and its feeder streams have been undertaken by State and local agencies, conservation groups, university departments, and private citizens.

Recently, U.S. EPA awarded to the State of Massachusetts (Division of Water Pollution Control (DWPC) and Division of Environmental Quality Engineering (DEQE)) Section 314 Lake Restoration and National Urban Runoff Program (NURP) funds to develop a pollution-related lake management program for Lake Quinsigamond. Both projects are currently underway. The DWPC is conducting the Section 314 diagnostic study. In January 1980 the DEQE solicited engineering services to prepare the NURP stormwater management plan for Lake Quinsigamond. Environmental Design & Planning, Inc., Cambridge, Mass. was awarded the overall engineering study. Meta Systems, Inc., Cambridge, Mass., a subcontractor, will perform the water quality impact modeling analysis.

LAKE QUINSIGAMOND NURP PROGRAM

The objectives of the NURP project for Lake Quinsigamond are as follows:

1. Develop an overall framework control strategy and plan for mitigating the impact of nonpoint source pollution on Lake Quinsigamond to achieve/maintain class B water quality;
2. Focus on and develop reasonably detailed engineering information at several catchment sites supportive of implementation of stormwater treatment/control demonstration facilities as part of the continuing 314 program effort;
3. Develop a calibrated methodology for estimating causal relationships between pollutant emissions and water quality impacts for continuing planning, control, monitoring efforts; and
4. Develop a sound data base of quantified land use/emission pollutant loadings, rainfall/imperviousness/runoff characteristics, and effective control/treatment alternatives that can be input into the NURP data files as well as provide information for similar studies in the New England region.

The work program for the Lake Quinsigamond NURP project is shown in Figure 2. The 314 program is in concurrent operation but is not described for the sake of brevity. A short description will be presented of only the stormwater measurement and control program formulation tasks with emphasis on settleability experiments and potential solids separation devices.

Task 1: Stormwater Measurement Program

The overall stormwater measurement program consists of three parts. The first program uses automotive equipment to measure flow/water quality at five locations within the Lake Quinsigamond watershed for 20 storm events. This information is meant to better define the land-use emission factors used in the runoff models. The second program consists of manual grab sampling at a number of secondary sites concurrent with the primary program. The aim of the secondary program is to obtain auxiliary data at other locations in the watershed. The third and final program entails obtaining, during storm events, large samples (151.4) liters of runoff at several key measurement locations and performing settling column tests. These tests will be used to define types of realistic controls in the watershed. The settling column tests will help to define the relative fraction of grit/easily settleable versus light colloidal material. Nutrient analyses will also be performed as part of the settling analysis so that relative fractions of nutrients attached to particles of differing sizes (settling velocities) can be ascertained.

Stormwater Solids Settleability Characteristics

Efficient and rational designs of solids separator devices center on the knowledge of the settling velocity characteristics of the solids particles and fractions to be removed from them. These devices may play an important role in the 314 implementation program for Lake Quinsigamond.

Since eutrophication of the lake is a major issue, the effectiveness of solids separating devices will also depend upon the partitioning of nutrients (especially phosphorus) between the dissolved and suspended fractions. Samples will be collected to permit measurement of each fraction. Because of the distribution of particle sizes and the general tendency for smaller (less easily removed) particles to contain/absorb greater quantities of phosphorous per unit mass, settling tests and evaluations of solids' concentrating devices should include direct phosphorus measurement. The bioavailability of the sediment phosphorous phase will be assessed on some representative fraction of the samples using algal growth potential tests and/or extraction procedures.

Settling characteristics of urban runoff are difficult to determine accurately using conventional procedures because of the presence of both large (quick to settle) particles such as sand and grit, and small, light fractions (long settling times). Conventional procedures such as hand-operated rotation and stirring or using compressed air for pre-mixing create undesirable characteristics including solids degradation, incomplete mixing, and generation of eddies and currents. A U.S. EPA study, "Characterization of Urban Runoff Settleability Characteristics," describes a new method developed by Environmental Planning & Design for obtaining representative and accurate characteristics. The state-of-the-art column is shown in Figure 3.

The concept encompassed steady-state, bi-directional rotation coupled with flow stators inside the cylinder to facilitate mixing. Two electric motors were wired through variable speed controllers to offer a high degree of uniformity and flexibility to the mixing process. The flow stators can be conceptualized as minimum disturbance deflectors assisting mixing by developing uniform, low velocity currents opposing the centrifugal forces generated by axial rotation. This balance of forces was considered an attempt to gain uniform distribution of solids across a cross-section of the column. The longitudinal rotation was the mechanism by which the solids would be dispersed throughout the horizontal axis of the column. The device has been used with artificial media of known particle size and seems to closely replicate theoretical settling rates. Comparative investigations using the new device and conventional methods such as air diffusion and plunger mixing showed significant differences in settling velocity curves for the same sample of combined sewer overflow.

The importance of settleability information for rational design of solids separating devices is depicted in Figure 4. Combined sewer overflow samples (151.4 liters) were obtained from three locations in Dorchester and were analyzed using the new approach. Swirl regulators can be designed to remove particles with settling velocities exceeding 0.15 cm/sec. Complete removal of grit (5 cm/sec.) can be expected using a properly designed swirl. The three figures show a range of partial solids removal and the two areas of no removal (A) and complete removal (B).

Task 2: Develop Control Program

Pollutant removal levels required to allow the lake and each of its net urban drainage tributaries achieve the water quality criteria established in the water quality goals for the lake defined in Task 2, will be defined by interactive catchment area emission/water quality input analysis. Emission modeling approaches will be used to roughly ascertain hydrologic design criteria such as design storm capture volume/rate/frequency coupled with expected pollutant loadings. The settleability results will be used to further fractionate controllable pollutant loads by class of treatment/control devices, i.e., what can be expected by solids separators such as Swirl/Helical Bend regulators versus removable loads by street sweeping, and microstrainers. In-situ lake treatment control techniques will also be investigated if the required removal of solids/organics/nutrients cannot be feasibly attained by emission control of pollutants entering the lake.

PHYSICAL TREATMENT SOLIDS SEPARATION DEVICES

Physical treatment alternatives are primarily applied for removing floatable, settleable, suspended solids and their associated pollutants from wastestreams, and are particularly important to stormwater and combined sewer overflow treatment. Physical treatment systems have demonstrated a capability to handle high and variable influent concentrations and flow rates and operate independently of other treatment facilities, with the exception of treatment and disposal of the sludge/solids residuals.

Swirl Regulator/Concentrator

The Swirl concentrator has demonstrated the capability to handle high and variable influent concentrations and flow rates with relatively high removal efficiencies. (See Figure 5-A.) This device can be designed to completely remove sand and grit, partially remove (40 to 60 percent) lightweight settleable particles, and substantially remove (45 to 80 percent) floatable solids at a fraction of the detention time (1 to 2 minutes) normally required for conventional sedimentation. Swirls are designed for hydraulic loading rates ranging from 37,850 to 151,400 liters/m^2/day, depending on the application. The device is perfectly suited for treating intermittent discharges (wet weather runoff) containing both settleable and floatable pollutants.

Helical Bend Regulator/Concentrators

Helical bend regulator/concentrators have been modeled, and design criteria as well as comparative cost evaluations have been developed and are presented in handbook form. Helical bends appear practical as in-line regulator devices commensurate with swirl. (See Figure 5-B.)

West Roxbury Swirl/Helical Bend R&D Facility

A major U.S. EPA effort is underway involving the design, fabrication, installation, and operation of two full-scale state-of-the-art stormwater pollution abatement devices. A treatment complex consisting of a swirl and helical bend regulator solids separator are being tested, side by side, for their ability to remove stormwater and simulated combined sewer pollution loads from a 65 hectare catchment area situated in West Roxbury, Mass., tributary to the Charles River. Environmental Planning & Design has designed, shop-fabricated, field-installed, and is monitoring wet weather pollutant removal effectiveness of the two devices over a 1-year period.

A plot plan of the facility is shown in Figure 8. The design flow for each unit (based on 3-week recurrence interval) is 6 cfs. Both units can be driven up to peak discharge of 18 cfs each. Discharge into each unit is evenly split using motor-activated bottom-opening sluice gates. Foul sewer underflows from both units containing the removed pollutants are flow controlled at 3 percent of the unit design flow by Hydrobrakes. These devices provide nearly uniform discharge under fluctuating lead conditions. No clogging problems have been experienced to date. Pertinent dimensions of the swirl are as follows: diameter — 3 meters, height — 1.4 meters, influent/effluent (clear) diameter — 0.6, and foul sewer effluent — 0.3 meters. The helical bend is 20 meters long and is 1.3 meters high; piping sizes are similar. The evaluation program commenced late in 1979 and will continue to the end of 1980. Pollutant removals to date approximate primary treatment.

It is believed that the swirl and helical bend flow regulator/solids separators will be very useful to communites as inexpensive, maintenance-free tools for combating stormwater pollution problems. For storm drain systems these devices can be installed on separate storm drains before discharge and the resultant foul underflow could be stored in relatively small tanks since concentrate flow is only a few percent of total flow.

In another approach, devices such as the swirl degritter or sedimentation basins may be used to provide final dewatering of the concentrate underflow suitable for disposal. Stored underflow could be later directed to the sanitary sewer for subsequent treatment during low-flow or dry-weather periods, or if capacity is available in the sanitary system, the foul underflow may be diverted without storage. This method of stormwater control would be cheaper in many instances than building huge holding reservoirs and it offers a feasible approach to treating separately sewered urban stormwater.

AN EXAMPLE OF URBAN WATERSHED MANAGEMENT FOR IMPROVING LAKE WATER QUALITY

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ABSTRACT

Many investigators have identified the urban environments as those producing high levels of water pollutants relative to other land uses. In a 55 hectare (136-acre) urban watershed in Orlando, a stormwater system discharges to an 11-hectare (27-acre) lake. The lake water quality is characterized by frequent algal blooms, odor, and in general, reduced human activities. The lake is one of the focal social areas of the city. Previously reported work on algal assays, bottom mud inactivation, and trophic analysis indicated that a mass of phosphorus should be removed to reduce algal blooms and improve the general aesthetic appearance of the lake. Lake water quality and stormwater impacts not previously published are presented in this paper. Stormwater runoff pollution mass and concentrations were estimated from a hydrograph related and composite sampling program. The average loadings and concentrations were compared to national data. A wide range of values was noted among storm events. Stormwater management procedures were established based on the runoff sampling program and a target reduction of phosphorus and metals. Estimates for the cost and benefits of the abatement program were completed. Management of stormwater for the removal of phosphorus was accomplished by diversion for retention of the first flush of pollutants. The efficiency cost curves were estimated from field performance data. For average yearly removals over 80 to 85 percent per year, these curves reflect rapidly increasing cost. Below 80 to 85 percent linear curves were typical.

INTRODUCTION

Stormwater may be a significant source of surface water pollution in urban areas (Weibel, 1969; Wanielista, 1977; Yousef, 1980). Lake impacts have been and continue to be studied on an international level. There exists in the United States a National Eutrophication Research Program (Gakstatter, 1975) and an international program with U.S. participation (Rast, 1978).

This paper documents stormwater impacts on an urban lake. The impact was first defined by visual observation. In a U.S. Government funded 208 program (East Central Florida, 1978), stormwater was reported to be the major pollution source. There were no point sources of industrial or domestic wastewaters. Thus, an investigation of the stormwater impacts was initiated and the results are reported here and elsewhere (Yousef, 1980). It was necessary to estimate stormwater composition, mass loadings, and impacts to determine a combination of management practices.

Evaluations of stormwater management practices have been completed prior to this work, such as those for urban areas (Field, 1977), and others (Wanielista, 1978). However, the critical relationship between a management practice and receiving water quality has not been well documented, except for some dissolved oxygen responses in rivers. This work aids in evaluating stormwater management practices to reduce impacts on lakes.

WATERSHED AND LAKE CHARACTERISTICS

The drainage area studied here is the Lake Eola watershed located in central Florida within the city of Orlando. The stormwater system is separate from the sanitary sewage system. The stormwater system drains a watershed of approximately 55 hectares (136 acres), composed of 31.7 hectares (78.2 acres) of commercial and 23.5 hectares (57.8 acres) of residential areas discharging to an 11-hectare (27-acre) lake. In addition, 4.5 hectares (11.2 acres) of parkland surrounds the lake. The watershed area was determined from storm sewer drawings and visual observation during rainfall events. Streets and parking lots comprise approximately 16.6 hectares (41 acres) of the watershed within a total of about 29.5 hectares (73 acres) of impervious lands. The pervious area is only 9 hectares (22 acres), most of which is in the residential areas.

Thirty-five parking areas discharge onto the street surfaces. Their total area is about 10 hectares (24 acres). These parking areas were identified as possible areas for management of stormwater. Since runoff waters discharge to the land-locked lake, one of the parking lots was designated a sampling location for runoff waters.

The 11 hectare (27 acre) lake is known for its picturesque setting. Its picture is the logo for the city of Orlando. It was once a natural lake, and historical records indicate that surface waters did not discharge from the lake. The level of the lake is usually maintained between 26.5 meters (87.0 feet) and 27 meters (88.5 feet) above sea level. Physical characteristics of the lake are shown in Table 1. It is a shallow lake with a mean depth of approximately 3 meters; about 73 percent of its volume is located within the 0 to 3 meter frustrum layers. Most of the 5,000 urban lakes in central Florida have similar physical characteristics.

BENEFIT

The benefits of the lake and its surroundings are evident, but difficult to quantify. The lake is a city focal point for residents and tourists, with frequent music concerts, arts/crafts shows, a children's park, and relaxation areas. The land values of property surrounding the lake bring top value because of its location. Lake Eola is one of the main reasons for the economic health of the downtown area. Estimates of the dollar benefits from lake activity are presented in Table 2.

Table 1. — Physical characteristics for Lake Eola, Florida.

Parameter	Quantity
Approximate surface area	109,270 m^2/11.0 hectares/(27.0 acres)
Approximate volume	3.30 X $10^5 m^3$/(8.73 X 10^6 gallons)
Mean depth	3.20 m/(9.92 ft)
Maximum depth	6.8 m/22.3 ft)
Length of shoreline	1417 m/(4650 ft)
Shoreline development	1.21
Volume development	1.72
Average height above sea level	26.8 m/(88 ft)

Table 2. — Estimated Lake Eola benefits.

Activity	Approximate frequency/year	Approximate people-visits/year	$/year
Music concerts[1]	35	87,500	262,500
Arts/crafts[2]	3	60,000	300,000
Tourist visits[3]	Constant	180,000	90,000
Fish-a-thons[1]	3	3,000	9,000
Food concessions[2]	Constant	---	100,000
Paddle boats[2]	Constant	5,000	20,000
Children's park[1]	Constant	125,000	187,500
Relaxation/aesthetics[1]	Constant	200,000	600,000
Jogging[1]	Constant	50,000	150,000
Land value[4]	Constant	---	600,000
	TOTALS	710,500	2,319,000

[1] Based on estimated attendance and an expenditure of $3 per person visit.
[2] Based on concession money received by the City of Orlando and an estimated attendance.
[3] Greyline of Orlando estimated visits as a portion of a larger tour.
[4] Based on lakefront vs. non-lakefront property taxes.

LAKE IMPACTS

This section summarizes the lake impact work completed to date. A more complete report is published elsewhere (Harper, 1980; Wanielista, 1980). Visual observation and analytical data reveal that Lake Eola has persistent algal blooms virtually year round. Populations of the macroscopic algae, *Chara*, and the filamentous green algae, *Spirogyra*, covered up to 30 percent of the lake surface area during the summer rainy season.

Lake Eola water was found to be somewhat alkaline with pH ranging from 8.4 to 9.5. Measurements of pH in Lake Eola indicate the rate of algal production. The average annual value and the measured range of values for pH, chlorophyll *a*, inorganic and organic carbon, and Secchi disk transparency are shown by Table 3.

The average values shown in Table 3 can be compared to values reported in the literature for eutrophic lakes, as indicated in Table 4. Lake Eola has some of the characteristics of an eutrophic lake.

Table 3. — Values for selected parameters measured in Lake Eola, Fla. between July 1978 and August 1979.

Parameter	Number of samples	Average value	Units	Standard deviation	Range of values
Chlorophyll *a*	64	25.4	mg/m±	8.8	9.0- 36.4
Organic carbon	67	10.9	mg/l	6.7	3.0- 29.1
Inorganic carbon	68	18.8	mg/l	6.4	13.8- 40.6
pH	57	8.9	--	---	8.4- 9.5
Secchi disk	32	106	cm	13.0	90 -120

Table 4. — The range of values for selected parameters in eutrophic lakes, as reported by Wetzel (1975).

Source	Chlorophyll *a* mg/m^3	Total P µg/l	Secchi disk cm	Organic carbon mg/l
Wetzel (1975)	10-500	10-30	--	5-30
EPA-NES (1974)	12	20	200	---

Concentrations of dissolved oxygen in Lake Eola, although usually at or above saturation near the surface, drop periodically during the spring and summer months to less than 1 mg/l in deep areas of 4 meters or more water column. Phosphorus from the bottom sediments was released up to a level of 250 mg/m^2 after 2 months of anoxic conditions (Marshall, 1980). This anaerobically released phosphorus has the potential for increasing water column phosphorus by 11.6 µg PO_4^{-3} - P/l, or about 50 percent of the average orthophosphorus concentration in the lake (23 µg/l).

When the concentration of orthophosphorus in Lake Eola was less than 0.10 mg/l, algal production was regulated by adding orthophosphorus alone. Above this concentration it appears that an excess of phosphorus was available, and algal growth was regulated by the N:P ratio. However, in most cases the concentration of orthophosphorus in Lake Eola water was below 0.04 mg/l, and algal production was most likely limited by the concentrations of added phosphorus alone.

In contrast to the enhanced algal growth conditions experienced during the summer rainy months, runoff entering the lake after prolonged periods of drought also produces several toxic effects on aquatic life in Lake Eola. Contaminants are allowed to accumulate within the watershed, and when a storm event occurs, the mass loading to the lake is many times larger than that experienced during frequent rainfall periods. This influx of toxic and oxygen-demanding wastes can kill many forms of aquatic life. Evidence of such a phenomenon was recorded in March 1979 when it rained after a dry period of 6 weeks. Concentrations of organic carbon as high as 400 mg/l were measured in stormwater runoff entering the lake during this rain. Two days later, dissolved oxygen concentrations had been reduced from saturation near the surface to 4 mg/l at a depth of 1 meter and to near zero below 2 meters. Numerous large-mouth bass averaging 2 to 3 pounds were found floating in the water, and large masses of dead filamentous algae had accumulated in thick mats over much of the lake's surface. During 1979 a total of six fish kills were reported. one dead bass fish (about 2 kilograms) floating at the surface was brought to the laboratory, processed and analyzed for metal concentrations in selected organs (heart, gall bladder, liver, stomach) and flesh. From the limited data available it appears that nickel and lead concentrated in the gills, iron concentrated in the heart, and zinc and copper concentrated in the liver. However, at this time, it is not known that these metals were directly responsible for the fish kill.

A pathogen isolation study was conducted over 1 year. One hundred twenty-nine water and sediment samples were collected. Fourteen were composites of runoff, 32 were bottom samples, and 83 were lake water. *Clostridium* was isolated from the bottom sediments of the lake and *Salmonella* was isolated from the lake water samples.

There are domestic ducks in the Lake Eola waters and park areas. On the average, they number approximately 20, with decreasing populations noted over the past 5 years. The population decrease is believed to be caused by *Clostridium* botulism. Microbiologists at the Orange County Pollution Control (Adams, 1977) have speculated that during site visits, gas production from the anaerobic sediments is increased in the summer months. This anaerobiosis promotes growth of the botulism organism which produces a toxin which, in turn, concentrates in the small insect larvae of the sediments. When ducks eat the larvae they can die.

Two dead ducks were sent to the State Veterinary Laboratory for autopsies. They were selected from among 35 dead ducks by the Humane Society. After autopsies, the Humane Society of Orlando reported that botulism caused the duck deaths in the lake (Orlando Sentinel Star, 1977).

STORMWATER

Stormwater pollutants and flow rate were first estimated by sampling stormwater relative to the hydrograph. Eight rainfall/runoff events were quantified in this manner. Next, a composite sampling program was completed with seven rainfall/runoff events. One major question was the percentage of dissolved pollution materials present in the runoff. The sampling program indicated that the dissolved nutrients and organics were approximately 50 percent or more of the total, while the dissolved fraction of lead was 20 percent. From the 15 runoff samples, estimates were made for average mass loading (as discharged) and average concentrations. These averages are shown in Table 5. Estimates of loading rates from both commercial and residential areas were calculated from the runoff studies.

The Lake Eola study loading site data are compared with the loadings of SWMM/level I analysis (Heaney, 1976) and other national data (Wanielista, 1979). The suspended solids and BOD data (Table 6) appear to reasonably agree. However, total nitrogen data are higher in the Lake Eola watershed. Possible reasons are that the residential areas should be classified as commercial areas when considering loading rate data, the landscaping maintenance places an additional nitrogen load, and the heavy rainfall (130 cm) is greater than the national average. Most likely, a combination of these reasons caused the increase.

Table 5. — Concentration and loading rate runoff summary (hydrograph related and composite sampling programs).

Parameter	Sample size (storms)	Mass loading range (kg/ha-yr)	Averages* Loadings kg/ha-yr	Averages* Concentration mg/l
Suspended solids	14	470 — 2,368	991.0	131.0
Volatile suspended	7	234 — 610	538.0	71.0
NVSS	7	76 — 587	453.0	60.0
BOD_5	8	40 — 315	98.0	13.0
COD	6	130 — 1,776	711.0	74.0
TOC	13	53 — 2,572	946.0	99.0
TKN	10	10 — 87	32.0	3.3
Ammonia-N	12	0.2 — 10.4	4.1	0.43
Total phosphorus	14	1.8 — 16.4	4.8	0.48
Zinc	9	1.2 — 5.5	3.7	0.38
Cadmium	9	0.09 — 1.0	0.28	0.03
Arsenic	8	0.17 — 1.76	1.02	0.11
Nickel	9	0.06 — 0.54	0.28	0.03
Copper	9	0.12 — 1.39	0.68	0.07
Magnesium	8	2.58 — 31.25	9.86	1.03
Iron	9	2.9 — 16.46	9.52	0.99
Lead	9	1.1 — 9.5	4.26	0.44
Chromium	9	0.07 — 0.51	0.25	0.03
Calcium	9	99.7 — 487	308.0	32.10

*Both commercial and residential

Table 6. — Loading rate comparisons.

	SS	BOD_5	TOC	TN	TKN	PO_4-P	TP
Lake Eola							
Commercial	1,076	196	1,167	32.0	27.8	1.7	3.5
Residential	827	87	757	40.5	36.1	3.1	6.2
+SWMM/Level I							
Commercial	1,255	181	--	16.7	--	4.3	--
Residential	922	45	--	7.4	--	1.9	--
++National averages							
Commercial	941	97	--	14.5	--	--	3.0
Residential	470	39	--	6.6	--	--	2.0

+ Heaney, 1976
++ Wanielista, 1979

The commercial and residential land use pollution contribution to the total was estimated to be 98 percent for suspended solids, 96 percent for BOD_5, 95 percent for total organic carbon, 94 percent for TKN, and 91 percent for TP. The total contribution was defined as the sum of stormwater, atmosphere, and ducks living on the lake.

The sampling program and the lake impact work led to the following conclusions: (1) stormwater is the major source of lake related pollution; (2) phosphorus and other stormwater pollutants had to be removed; (3) sedimentation was possibly not the choice method for stormwater management because of the large percentage of dissolved pollutants.

TARGET PHOSPHORUS REDUCTION

The major question is to what degree should the bottom sediment and stormwater be treated to economically reduce the nutrient enrichment, fish and duck kills, and algal activity to an acceptable level? Using the trophic state models, a target reduction level of phosphorus loadings in the oligotrophic/mesotrophic level may reduce algal blooms. In addition, a chlorophyll *a* mean concentration of 7 μg/l may indicate a mesotrophic state. Table 7 illustrates the target level and the need for an approximate 90 percent reduction in phosphorus load and phosphorus concentration.

Table 7. — Target reductions.

Models	Before	Target Reduction Levels
Vollenweider	2.33 g-P/sq m/year	0.2 g-P/sq m/year
Dillon	0.49 g-P/sq m	0.05 g-P/sq m
Larsen-Mercier	0.48 mg/l	0.05 mg/l
OECD/chlorophyll[1]	269 mg-P/m^3	70 mg-P/m^3

[1] Reduction corresponding to a **chlorophyll *a*** of 7 μg/l (Gakstatter, 1975)

In the National Eutrophication Study total phosphorus concentration of less than 10 μg/l in the water column was noted as a target reduction to classify lakes as oligotrophic. A combination of stormwater treatment and bottom sediment inactivation may produce a water column concentration of less than 10 μg/l. The bottom sediments were estimated to contribute 11.6 μg/l of the average water column concentration of 23 μg/l .

STORMWATER MANAGEMENT SELECTION

Each stormwater management practice that could be defined in terms of cost and efficiency and was practical for the watershed was evaluated for stormwater control. The selection of the best combination of practices was based on those which meet cost and efficiency constraints. With many practices and control locations within the Lake Eola watershed, the selection of the best combination (least cost) could be aided by a computer analysis.

Cost-efficiency curves (present value dollars versus removal quantities) were developed for each subwatershed of the Lake Eola watershed. Removal efficiencies from the literature (Field, 1977; Lager, 1977) and local 208 programs (Calabrese, 1977; Wanielista, 1979) were used. These 208 efforts in the central Florida area had defined the efficiencies and costs for diversion/percolation basins, swales, underdrains, and vacuum sweeping nonpoint source management methods. In the highly impervious urban areas, the cost of land is expensive, and land intensive activities (detention and retention basins) are sometimes not aesthetically pleasing. Thus, street sweeping, diversion with retention underground, and catchbasin cleaning appeared probable for urban areas. Dutch drains, rooftop storage, coagulation, filtration, and concentrators were other management methods under investigation.

These formed the basis for determining optimal combinations of practices. This was accomplished using a computer program written for this work. A linear programing network routing model was incorporated. The cost-efficiency curves were estimated by "piecewise" linear approximation (Calabrese, 1979).

One limitation on stormwater control was the use of private property. Thus, it was decided to do all management within the city right-of-way. The alternatives considered for management of the stormwater

1. Diversion of stormwater to the sanitary sewer system for treatment.
2. Street cleaning by both broom and vacuum sweepers.
3. Diversion of stormwater into percolation basins.
4. Conversion of inlets to catchbasins.
5. Coagulant addition with sedimentation.
6. Silt removal from lake, and drawdown every 5 years.
7. Natural "living filter" treatment.
8. Fabric bag filters.
9. "Best" combination of any or all of the above alternatives.
10. Diversion of stormwater into infiltration trenches.
11. Others, such as sand filtration, swirl concentrators, and diatomite filters.

The first alternative was eliminated because it was not considered as a general solution for other areas and it required replacing over 7,000 meters of sanitary sewer lines, thus the capital cost of pipe and pumping stations was over $600,000. The number 11 alternatives were not developed because of lack of technical data on performance (cost vs. efficiency data) in a storm sewer system.

Before the percolation alternatives could be considered the infiltrative capacity of the soils was estimated. This was done by defining the type of soils and the location of the water table. The water table is at least 2 meters (6 feet) below the ground surface for a ground elevation of 29 meters (95 feet) or higher. Borings close to the lake indicate the water table is near elevation 27 meters (88 feet). In addition, sandy soil is available to about 6.5 meters (20 feet) below ground level. Percolation of stormwater is possible for parking lot and street drainage for those areas whose ground elevation is above 29 meters (95 feet).

All alternatives were evaluated in terms of estimated cost and yearly pollutant removal efficiencies. The cost for the natural living filter areas were estimated from local contractors and the city of Orlando records. The vegetation selected is native vegetation and has been used in other lakes. All other cost data were obtained from recently bid sewer projects.

The solution selected was based on minimum present value cost and maximum removal efficiencies. The fabric bag alternative had a lower capital cost, but poor removals relative to other alternatives. The locations of the best management practices were near parking lots and immediately before lake discharge. Parking lot and street diversion were designed to percolate the first 0.6 to 1.25 cm (1/4 to 1/2 inch) of every storm chosen. This results in a removal efficiency of 90 percent on a yearly basis. The resulting capital cost for stormwater management and lake restoration was approximately $6,250/hectare ($2,500/acre) of watershed.

CONCLUSIONS

Based on citizen concern and historical water quality data on fish and duck kills, oxygen depletion, and algal blooms, it was evident that the factors causing the water quality impact had to be identified. Trophic state analysis indicated that the lake was estimated as eutrophic. In laboratory tests, algal productivity was related to stormwater. Also, the bottom sediments were shown to contribute to the phosphorus concentration in the water column.

Based on the runoff quality and quantity data with lake limnological data, an implementation plan for stormwater management was developed. Since phosphorus is most likely the limiting nutrient, it will be controlled. The two major sources of phosphorus are stormwater and lake bottom mud recycle. By reducing stormwater phosphorus mass, re-stocking, littoral zone planting, and coagulant coverage of bottom muds, it is predicted that the effects of stratified conditions (anaerobic) will be minimized and algal blooms will be reduced.

The stormwater management will be done by diversion/percolation of parking lot runoff and limited street runoff (approximately 17 hectares, (40 acres)). In addition, those areas not managed with this method will be diverted into trench storage for infiltration into the lake (approximately 28 hectares, (65 acres)).

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LAKE RESTORATION BY EFFLUENTS DIVERSION IN FRANCE

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ABSTRACT

Sewage diversion has been applied by French authorities to protect two main lakes against eutrophication. The first one, Lake Annecy, is 27 square kilometers wide and showed signs of deteriorating after World War II because of domestic sewage. A pipe was thus constructed to collect these effluents all around the lake for treatment in a downstream purification plant. The construction started in 1962 and was completed in 1976. The response to reduced nutrient influx has been an obvious lake oligotrophication. The second lake, Lake Le Bourget, is 44 square kilometers in area. Eutrophication had been deteriorating its water quality for almost 30 years. To reduce the nutrient influx, essentially urban generated, most of the treated sewage was collected and diverted outside the drainage basin into the Rhone by a 12.3 kilometer-long tunnel excavated through a mountain. The diversion operation started in 1979 and no improvement has been observed until now.

INTRODUCTION

Diversion of sewage from a lake that is being enriched by the algal nutrients present in wastewater is the radical solution French authorities have found for the eutrophication problem in two main water bodies: Lake Annecy and Lake Le Bourget. These lakes are located in the same alpine area, less than 100 kilometers southwest of Geneva (Figure 1). Their main physical characteristics are given in Table 1.

Table 1. — Lake Annecy and Lake Le Bourget: main physical characteristics.

	Lake Annecy	Lake Le Bourget
Altitude above sea level (m)	446.5	231.5
Surface area (km^2)	27.04	44.62
Maximum depth (m)	64.7	145.4
Mean depth (m)	41.5	81.14
Volume (km^3)	1.1235	3.6203
Mean residential time (months)	44.0	36-48
Drainage area (lake excepted)(km^2)	251.0	560.0

LAKE ANNECY AND ITS PERIPHERAL COLLECTOR

Lake Annecy (Figures 2,3) began showing real signs of eutrophication during the decade after World War II, an evolution considered likely since 1943. Professional and amateur fishermen were perturbed at the marked decline of salmonid populations such as omble and trout. They rightly associated this with the rapid efflorescence of phytoplankton blooms resulting from a higher concentration of nutrients. Furthermore, the authorities responsible for monitoring the drinking water supply drawn from the lake were more worried by laboratory reports of rising bacteria counts, especially during the summer. The people really concerned by this deterioration were, of course, those living in nearby towns and lakeside communities, a population totaling 75,000. This group called in authorities to assess the state of the lake. It embarked simultaneously on a public education campaign, aimed first at the leaders of the lakeside communities.

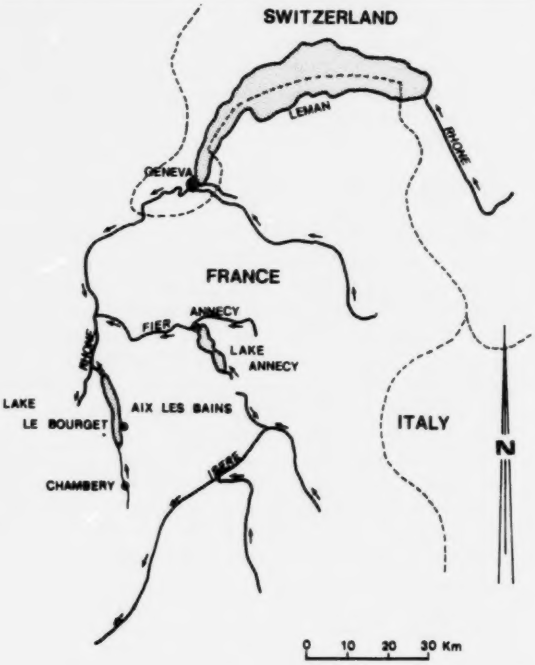

Figure 1. — Location of Lake Annecy and Lake Le Bourget.

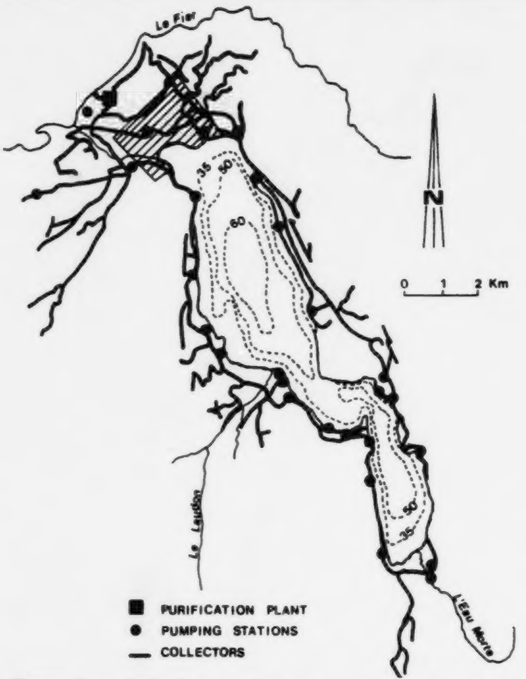
Le Fier
35
50
60
50
35
N
0 1 2 Km
Le Laudon
L'Eau Morte
■ PURIFICATION PLANT
● PUMPING STATIONS
— COLLECTORS

Figure 2. — Lake Annecy: bathymetric map and sewerage system.

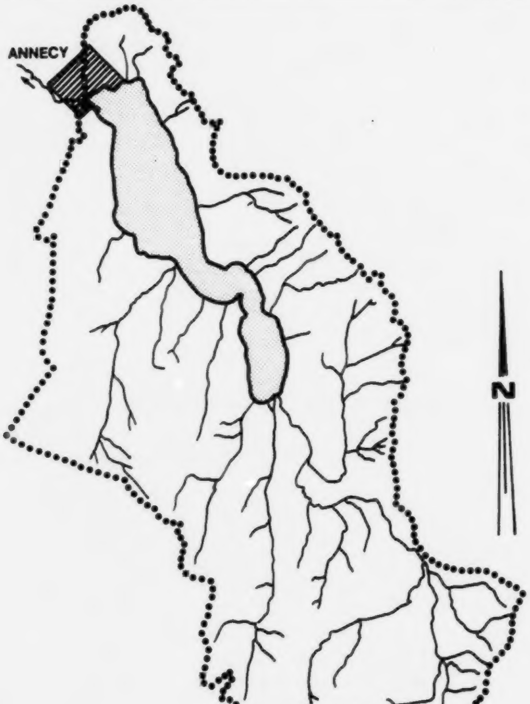
ANNECY
N
0 1 2 3 4 5 Km

Figure 3. — Lake Annecy: map of the drainage basin.

The situation appeared so serious that two technical solutions were proposed. One was a chain of purification plants for all wastewater then being discharged into the lake. The other involved collecting this water in main collector pipes running all around the lake and leading to a single large treatment plant. This plant would in turn discharge into the Fier, a tributary of the Rhone, below Annecy. The second alternative was chosen.

Initially, a syndicate of eight communities was formed in July 1957 to cleanse the lake. Today, the group includes 21. The main difficulty was persuading these communities, set in their traditional way of life, to commit themselves to expenditures likely to burden taxpayers for the foreseeable future. The construction began in 1962 and was completed in 1976.

A schematic diagram of the sewerage system is shown in Figure 4. Table 2 gives the repartition of these 280 kilometer pipes collecting wastewater from 103,000 inhabitants (Figure 2). The treatment plant capacity is today 135,000 gallons per inhabitant; it purifies 40,000 m^3/day^{-2} by a two-step process which is both mechanical and biological (activated sludges.) In addition, there is a 120 ton/day^{-2} composting plant and a 50 ton/day^{12} incineration plant for dealing with domestic and industrial waste. A total investment of 110 million French francs (1975) has been realized but this represents only half the total amount projected for satisfying the demand predicted for the year 2000.

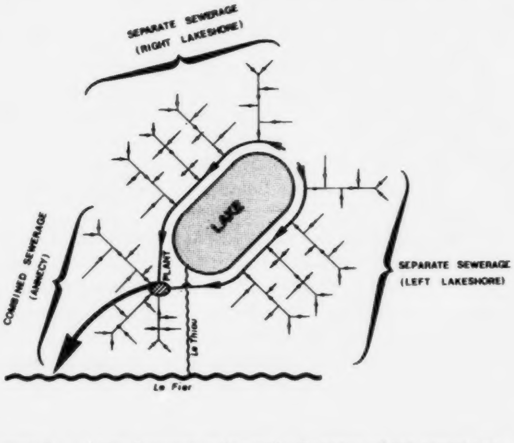
SEPARATE SEWERAGE
(RIGHT LAKESHORE)
LAKE
PLANT
COMBINED SEWERAGE
(ANNECY)
SEPARATE SEWERAGE
(LEFT LAKESHORE)
Le Thiou
Le Fier

Figure 4. — Lake Annecy: schematic diagram of the sewerage system.

Table 2. — Lake Annecy: technical characteristics of the sewerage system.

	Principal collector			Secondary collector	
	Length (km)	Diameter (mm)	Pumping Station (Number)	Length (km)	Pumping Station (Number)
Separate sewerage (Right lakeshore)	19	800-200	4	64	10
Separate sewerage (Left lakeshore)	28	700-200	7	132	9
Combined sewerage (Annecy)	37				

A few years of recuperation has improved the water quality, first in the epilimnion of the northern basin, the first to be protected. Nutrient concentrations are now at oligotrophic levels, transparency is restored, phytoplankton is dominated by diatoms, and the water has again become drinkable after only a limited filtration and ozonation process. However, a new problem recently appeared: The quantitative decrease in the fish catch may be related to restoring the lake's oligotrophic state.

LAKE LE BOURGET AND ITS DIVERSION TUNNEL

Lake Le Bourget (Figures 5,6) is the largest lake in France. It was described as oligotrophic in 1947. But around 1952-1955 it became clear that the lake was deteriorating because profuse algal blooms appeared. First localized along the most populated shores, they finally invaded the whole lake. Transparency decreased simultaneously with hypolimnetic oxygen and salmonid populations.

Preliminary investigations showed the nutrient input must be reduced by 95 percent to restore the lake. The construction of orthodox treatment plants appeared insufficient and tertiary treatment by chemical or biological nutrient elimination seemed to be too unreliable. Therefore, it was decided first to enlarge the collector network to a 95 percent capacity; secondly, to treat all collected effluents (primary and secondary treatment); and thirdly, to divert all treated sewages outside the drainage basin.

Two syndicates were created. Between 1960 and 1973, the two first steps were realized, especially in the districts of Chambery and Aix-les-Bains, whose effluents represented 85 percent of the point source input. This program has continued until now; a 400 kilometer sewer network collects wastes from more than 95 percent of the population and nonagricultural industries.

The third step began in 1973. Treated sewage from Chambery, Aix-les-Bains, and Le Bourget du Lac were collected and drained directly to the Rhone by a tunnel excavated through a mountain. Figure 6 shows the location of the different operations, Figure 7, a schematic diagram, and Table 3, their main technical characteristics. This sanitation action was reinforced

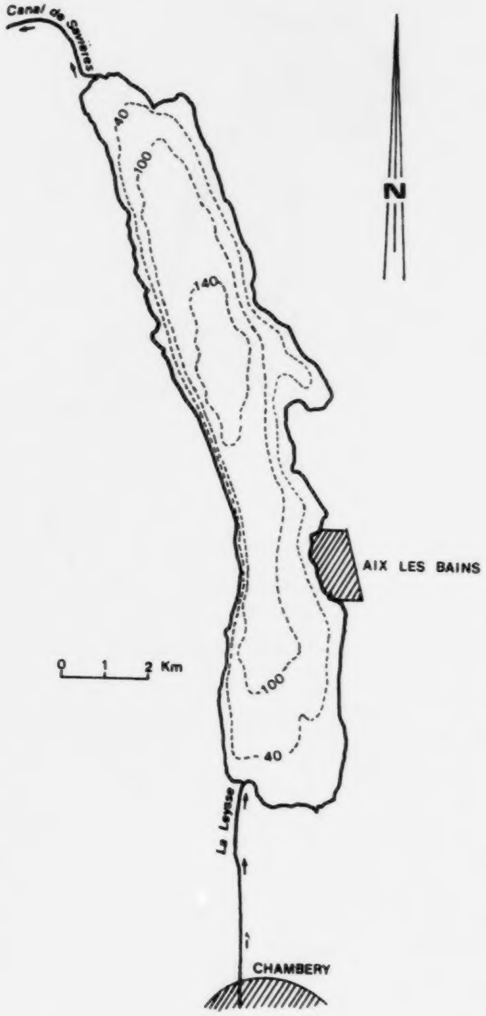

Figure 5. — Lake Le Bourget: bathymetric map.

Table 3. — Lake Le Bourget: technical characteristics of the diversion operations.

	Mains			Flows	
	Diameter	Cross Sectional area	Length	Type	Volume
	(mm)	(m^2)	(km)		($1.s^{-1}$)
Chambery — Le Bourget du Lac discharge pipe	1,200		8.2	by gravity	1,630
Aix-les-Bains — Le Bourget du Lac discharge pipe	600		7.6	by pumpage	580
Mont du Chat diversion tunnel		4.35	12.3	by gravity	$7,500.10^3$
Chindrieux — Rhone diversion pipe	400		5.2	by gravity	35

by an incineration plant, the biggest in France, and a mobile device for collecting floating detritus. A total of FF 280 million has been spent since 1965, 170 for the diversion. As this diversion functioning started only at the beginning of 1980, it is too early to observe any improvement.

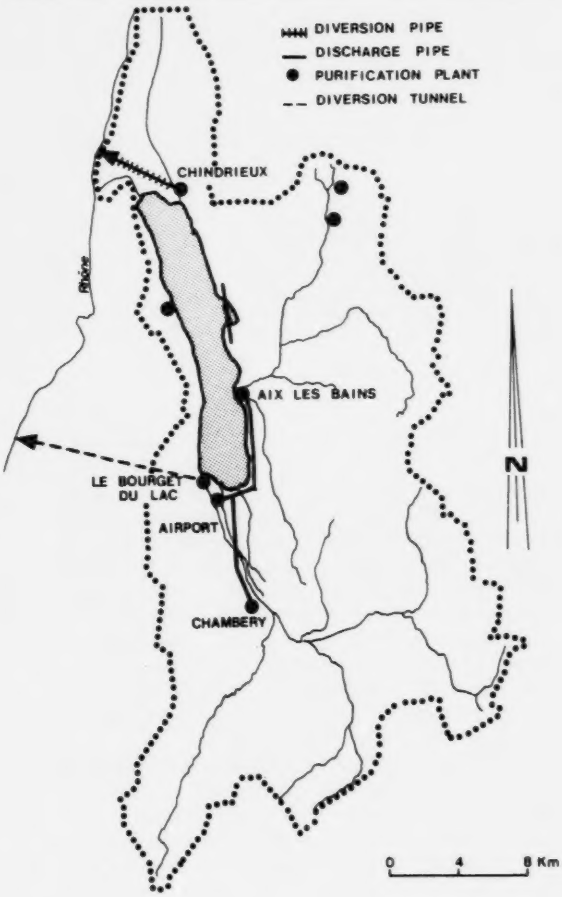
DIVERSION PIPE
DISCHARGE PIPE
PURIFICATION PLANT
DIVERSION TUNNEL
CHINDRIEUX
Rhône
AIX LES BAINS
LE BOURGET DU LAC
AIRPORT
CHAMBERY
N
0
4
8 Km

Figure 6. — Lake Le Bourget: map of the drainage basin and location of the diversion operations.

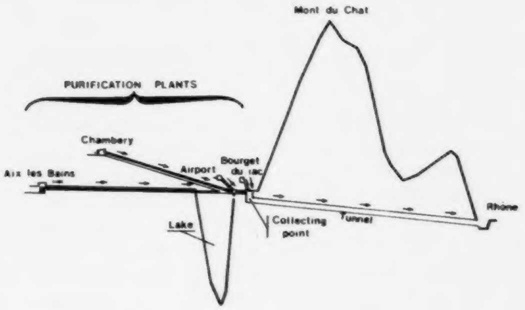
Mont du Chat
PURIFICATION PLANTS
Chambery
Aix les Bains
Airport
Bourget du lac
Lake
Collecting point
Tunnel
Rhône

Figure 7. — Lake Le Bourget: schematic diagram of the diversion operations.

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Documents concerning the Lake Le Bourget purification may be obtained from: Syndicat Intercommunal du lac du Bourget (SILB), 73000 AIX LES BAINS, France, Tel. 79/35.00.51.

Syndicat Intercommunal d'Assainissement de la Region de Chambery (SIARC),Rue Aristide Berges, 73000 Chambery, France, Tel. 79/69.58.69.

PHOSPHORUS BALANCE AND PREDICTIONS: LAKE CONSTANCE, OBERSEE

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ABSTRACT

Results obtained from a dynamic model for the phosphorus balance of Lake Constance are discussed. Accordingly, the eutrophication process is based upon an increase of the phosphorus loading from waste waters, especially from detergent phosphorus. When the planned sanitation measures are finished, the lake will not return to its original state. However, it is able to react quickly upon a decrease of the loading because of large sedimentation rates. It is pointed out that the model requires development: Consideration of the river waters and loadings entering deeper layers of the lake, a further division of the water body during stagnation, and the calculation of fluctuations of the lake's volume.

INTRODUCTION

During the last 5 to 10 years numerous models to judge the trophic states of lakes have been proposed (Schindler, 1978; Vollenweider, 1976; Schroeder and Schroeder, 1978). Morphological, hydrological, chemical, and biological parameters were combined, considering loads and concentrations of important elements or compounds, residence time of water, density of biomass, and production. Emphasis has been on comparing lakes differing in degree of pollution and describing their conditions. The reaction of lakes after change in a parameter can be derived, but there are uncertainties in the quantitative prognosis of the development of the lakes and of the chances of practical measures succeeding in a single case. This is more possible with a dynamic model fitted to the special lake. With such a model behavior of other lakes also can be understood.

This paper presents experiences gained with a simple dynamic model for the phosphorus budget of Lake Constance (Wagner, 1976a). With its aid the following questions should be answered:

1. How did the yearly phosphorus loading of the lake develop and from what sources?
2. What phosphorus concentrations are to be expected in the lake without sanitation measures?
3. How do these measures affect the phosphorus balance?
4. What has to be taken into account towards future development of the model?
5. Which phenomena need more research?

In the mid-1930's about 5 mg P/m^3 were measured and o-phosphate could not be found. Since the 1950's the phosphorus concentration has increased: slowly at first, then more rapidly. The consequences were increasing the density of biomass and the algal blooms, and decreasing oxygen in the hypolimnion of the summer stratified lake. O-phosphate always was present in the hypolimnion. Today, a research program is investigating loading from the tributaries, the wastewater treatment plants, the atmospheric precipitation, and the concentrations in the lake at several stations.

Lake Constance is used as a drinking water reservoir, for recreation, and by fisheries. In 1960 the International Commission for Water Protection was founded. It decided to treat plants with phosphorus precipitation and decided against a ring channel. The main reason for this decision was that a large catchment area would have had to be connected to the ring channel. The measures were started in the early 1960's, with building since about 1970 at great financial expense. Most of the treatment plants have begun to work after 1975. The sanitation program will end in the 1980's.

Estimations of phosphorus balance data were available on the mentioned phosphorus sources, on water discharge, concentrations in the lake, lake stratification, and statistics on turnover of polyphosphate, fertilizers, and on the development of the human population.

ROLE OF THE PHOSPHORUS SOURCES

Less phosphorus always leaves the lake than has been added (even after subtraction of the phosphorus within the suspended matter of rivers). This means that both particulate phosphorus and a large amount of the originally dissolved compounds remain in the lake. We can assume that the particulate fraction enters the sediment within a few months. *There are between 2,000 and 3,000 tons during high water years and less than 1,000 tons P/yr. during low water.* Suspended matter in the less polluted rivers Rhine and Bregenzer Ach *adsorb* o-phosphate from the lake water, while those of the other more polluted rivers *add* phosphate to the water (Wagner, 1976b). But altogether the effects of these processes seem to be compensated. In the case of Lake Constance it was more advisable to hold this particulate phosphorus *not* responsible for the eutrophication process. Therefore, the balance is restricted to the sum of all phosphorus compounds with the exception of particulate ones from the rivers.

These data were used to estimate the yearly phosphorus loadings from polyphosphate, sewage, precipitation of the catchment area and the surface of the lake, fertilizers and the geological formations for the years from 1930 to 1975 (Figure 1).

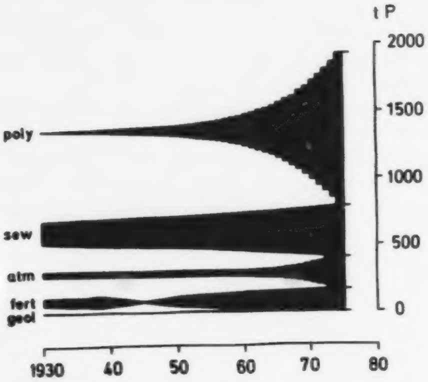

Figure 1. — Estimation of the yearly phosphorus loading of Lake Constance (without particulate compounds from rivers) and its separation into sources: poly = polyphosphate; sew = sewage without polyphosphate; atm = atmospheric precipitation; fert = fertilizers; geol = geologic formation.

PHOSPHORUS BALANCE AND PREDICTIONS

Before modeling, the following assumptions had to be made (Wagner, 1976a): (a.) The annual phosphorus loading for the present enters the epilimnion; (b.) the suspended material from the rivers deposits immediately; (c.) morphological particulars of the lake are not considered; (d.) by precipitation, sedimentation, and circulation phosphorus enters deeper layers of the water body; (e.) the different phosphorus compounds in the lake are interchangeable. The model works with monthly intervals; the waterbody is divided by the thermocline only. Inverse temperature stratification does not occur. Only one circulation period exists.

Input data include the monthly phosphorus loading and water discharge. Output data and fitting parameters have been mean phosphorus concentrations in the epilimnion, the hypolimnion, and at the end of the circulation period (March/April) of the total lake as well as the yearly load in the outlet of the lake. Using the coefficients in the terms for sedimentation rates from epi- and hypolimnion, the model has been adapted. These terms take over all downward phosphorus transports.

After adapting the data, the large amount of the yearly (calculated!) sedimentation rates of originally dissolved phosphorus compounds were noticed (Figure 2). Phosphorus may reach the sediment transported by skeletons of organisms, in the course of calcite precipitation (Rossknecht, 1980), adsorbed by sinking particles, or chemically precipitated.

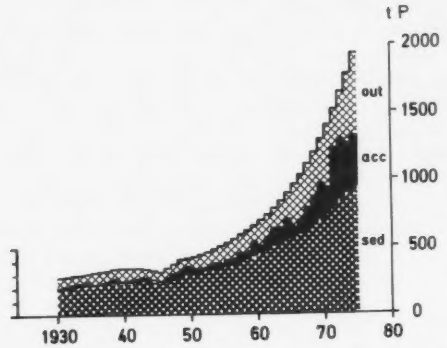

Figure 2. — Whereabouts of the yearly phosphorus loading of Lake Constance (without particulate compounds from rivers): out = outlet of the lake; acc = accumulated in the water body; sed = deposited in the sediment. In 1944/45 impoverishment in the water body (blank).

In the period of increasing phosphorus loading the budget was *never in a steady state*. During years with low water (1971-72) a large amount of phosphorus accumulated in the water body contrasted with a relative small amount during high water. Also changes in the yearly water discharge caused a non-steady state which equalized over a long period. This means that high water periods equal decreasing phosphorus concentrations in the lake; low water periods equal increasing concentrations.

After the rapid increase in loading, the phosphorus budget now seems to be in a steady state (temporarily?) and a concentration plateau has been reached. Though in the last years the densities of phytoplankton biomass did not develop proportionally to the phosphorus (self-shading? Buergi and Lehn, 1978), still phosphorus limitation seems to exist, at least during August/September. It can be said that a significant diminution of the phosphorus loading also decreases biomass and a recovery of the oxygen budget.

The simulation of steady states resulted in the following relations between yearly loading (without particulate P from rivers) and concentration at the end of overturn (total P March/April) in the lake:

2,000 tons P	135 mg P/m^3
1,500 tons P	90 mg P/m^3
1,000 tons P	45 mg P/m^3
500 tons P	17 mg P/m^3

The future turnover of phosphorus in the lake will lie within the observed ranges only as considered in the adaptation of the model. Therefore, prognosis based on a time extrapolation may be allowed. About 2,000 tons P/year were calculated for the mid-1970's. Without sanitation measures, more than 100 mg P/m^3 would have been expected (Figure 3) in the lake. Lake Constance has rather a long residence time (4.4 years). Nevertheless, the lake would quickly react to a decreased pollution: Within 10 to 15 years after total cessation of phosphorus input, phosphorus would disappear from the lake waters (simulated example) because of the great phosphorus uptake (Edmondson, 1979; Imboden and Gächter, 1978) and sedimentation. The rest of the loading — after the planned measures have been effected — amounts to about 1,200 tons P/year (without the suspended matter of the rivers). The resulting steady-state phosphorus concentration in the lake will then go down to 60 mg P/m^3. But a large increase in the phosphorus turnover in the catchment area will affect the lake again.

Today, the observed concentration of 80 to 85 mg P/m^3 (close to steady-state) corresponds to a simulated loading of less than 1,500 tons P/year. Apparently the measures have already decreased phosphorus. If these sanitation measures are not completed and development continues with more homes discontinuing septic disposal and connecting to the sewage plants, phosphorus will again increase. Information about this will be available after a current investigation has been completed. However, the effectiveness of treatment plants can be improved and the phosphorus content of detergents can be reduced.

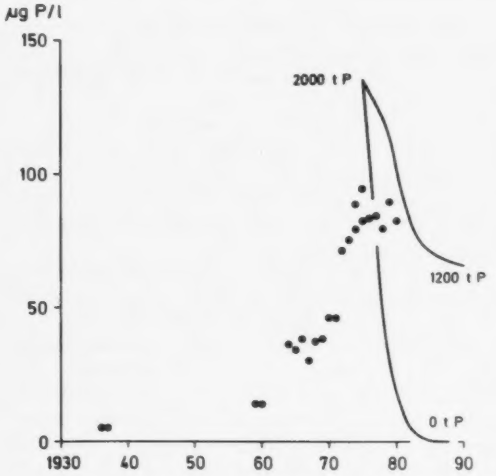

Figure 3. — Overturn concentrations of total phosphorus in Lake Constance; calculated steady state concentrations during a period of permanent loading of 2,000 tons P/Yr (without particulate compounds from rivers), decrease of this loading to 1,200 tons P/Yr within 5 years as an effect of measures or to zero after an Utopian total stop of phosphorus input.

DEVELOPMENT OF THE MODEL

The difficulties of getting data for a balance do not occur in the investigation of the lake itself but in the record of the seasonal variation of the loading to the lake. Experiences show that the load of suspended matter in the rivers, for instance, can be determined by special programs only. Also, particulate compounds should be separated from dissolved ones because of their different behavior. With respect to judging the success of sanitation, the known difficulties exist with determining the origin of the loads. The use of statistics is risky and the expense of chemical investigations all over the catchment area is rather high. So an attempt has been made to calculate and separate point source and diffuse loading of dissolved compounds with data only from the mouths of rivers (OECD, 1979). However, further separation of the suspended matter corresponding to its origin, also in the future, might be uncertain (interim sedimentation!).

The adaptation of the model with the aid of the chosen terms for sedimentation rates was satisfactory. However, a number of processes occurring in the lake had to be temporarily disregarded.

Meanwhile, it has been confirmed that phosphorus release rates from profundal sediments of Lake Constance (Obersee) are very small, because more than 1 mg oxygen always exists (Frevert, 1980). *It will not be relevant* to the phosphorus balance.

Also, the influence of the large concentrations of suspended matter during high water on the behavior of river water within the lake has been neglected too much. Investigation of the relationship between density of river water and seasonal course of temperature, suspended matter concentrations depending upon rain falls, and concentrations of dissolved salts have shown (Wagner and Wagner, 1978) that the densities of lake and river water differ significantly during a year. Large concentrations of suspended matter during high water (up to about 5 g/l, grain size median 10 / um! Wagner, 1976b) exceed the effects of temperature.

The water flow through the lake decisively influences the phosphorus budget. After summer stratification begins, the snow in the Alps starts to melt, carrying waters low in dissolved phosphorus. The melt water is cold and rich in suspended matter it pushes forward into deeper layers of the lake. The outflowing waters from the lake mainly originate from the warmer epilimnion, where the production of biomass has already started. Phosphorus uptake, sedimentation, and displacement of epilimnic waters result in a quick decrease of phosphorus at the surface of the lake during May (Figure 4). The reasons for the short-term maximum of total phosphorus in April/May are still unsettled, because the differentiation is difficult between phosphorus in plankton and in fine grained matter from rivers.

The metalimnion is an efficient barrier for the incoming river water. From May to October — with the exception of high waters rich in particles — river water remains near the thermocline above 50 meters. But from November to March it mainly flows into a depth of more than 50 meters. So phosphorus from the rivers is not available for phytoplankton until it is circulated to the upper layers. The epilimnic impoverishment in phosphorus during August/September certainly results from throttled supplies. Therefore, considering the fact that the largest portion of the dissolved and the particulate phosphorus compounds of the river waters mixes into the hypolimnion, a change for the better in modeling results can be expected. Instead of dividing the water body by the thermocline only, an additional sectioning (at least epilimnion, metalimnion, and hypolimnion) will be necessary. Besides, the large fluctuation of the water input also causes fluctuations of lake volume (and water level) which have to be considered by an additional hydrological model unit.

Finally, monthly calculations will be made of the relations between phosphorus and biomass, production and oxygen. Some of the numerous empirical model functions will be used, because modeling real processes is not yet possible. However, improvements can be made by using additional diverse connecting parameters and functions (e-functions, geometric functions, iterative determination of coefficients). *In any case the model should be as simple as possible.*

The model functions mentioned at the beginning have the incontestable advantage of simple handling. On the other hand, dynamic models need a long time for development and adaptation to given facts; this usually makes long-term operators necessary. Sometimes experiences and ideas disappear if such a specialist changes his employment. At universities such models are evolved, too. But there the long-term data sets and the experiences of the practical men are not readily available. In contrast on-the-spot-modeling frequently is not possible for different reasons. Only large institutions with a sufficient number of collaborators are able to connect both. At present in Germany a discussion of these problems is taking place.

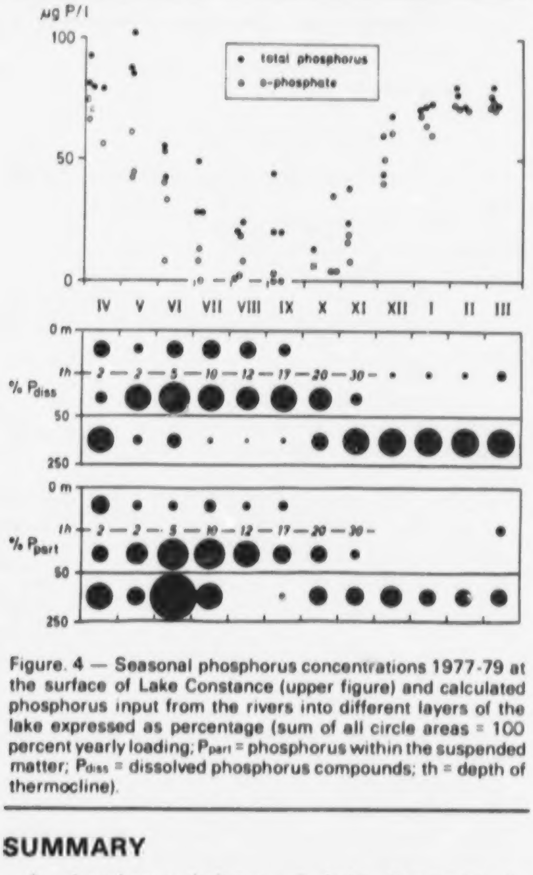

Figure. 4 — Seasonal phosphorus concentrations 1977-79 at the surface of Lake Constance (upper figure) and calculated phosphorus input from the rivers into different layers of the lake expressed as percentage (sum of all circle areas = 100 percent yearly loading; P_{part} = phosphorus within the suspended matter; P_{diss} = dissolved phosphorus compounds; th = depth of thermocline).

SUMMARY

A phosphorus balance of Lake Constance is calculated by a dynamic model. Data since 1935 are available. Loading after World War II was largely based on sewage (polyphosphate and feces). The main portion of the yearly loading enters deeper layers of the lake. Particulate phosphorus from the rivers seems not to influence the balance decisively. Today, the turnover concentration of total phosphorus is 80 to 85 mg P/m^3. Nevertheless, in August/September o-phosphates still seems to limit the algal production.

The model simulates steady states for different levels of loading. The main results are: Without sanitation measures turnover concentrations of total phosphorus of more than 100 mg P/m^3 can be expected. After the planned sanitation is finished, the lake will not return to its original state, for pollution from other sources remains too high. The concentration then will amount to about 60 mg P/m^3. The lake is able to respond rather quickly to a decrease in the loading because the sedimentation rates are high; they include the suspended fraction from the rivers plus more than 50 percent of the rest of the phosphorus loading.

To improve the phosphorus balance model, the supply of river water and phosphorus to the deeper layers of the lake and a further division of the water body during stagnation and seasonal volume variations ought to be considered.

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PREDICTION OF TOTAL NITROGEN IN LAKES AND RESERVOIRS

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ABSTRACT

The basic Vollenweider input-output model was adapted to predict total nitrogen concentrations in standing waters. Data from randomly selected group of lakes in the U.S. Environmental Protection Agency National Eutrophication Survey were used to develop the coefficients for the model, and data from a different group of lakes from the same survey were used for verification. The 95 percent confidence interval for predicting total nitrogen in a lake is from 41 to 255 percent of the calculated value for the best models. The same equations could be used equally well for natural lakes or artificial reservoirs.

Phosphorus and nitrogen have long been recognized as the two elements most likely to limit biological production in inland waters: thus, their cycles have been the subject of intensive research. An important advance was made by recognizing the importance of continuing nutrient inputs in the determination of trophic state (Vollenweider, 1968) and the development of input-output models for predicting nutrient concentrations on the basis of nutrient loading, lake morphometry, and hydraulic flushing rate (Vollenweider, 1969). Since that time, a number of empirical models have been developed to predict total phosphorus concentrations (Vollenweider, 1975; Kirchner and Dillon, 1975; Chapra, 1975; Jones and Bachmann, 1976; Larsen and Mercier, 1976; Reckhow, 1977, 1979; Canfield, 1979). Yet little effort has been expended on developing similar models for the other important element, nitrogen. The purpose of this study is to develop and test an input-output model for total nitrogen in natural and artificial lakes.

Unlike phosphorus with only one valence state in natural waters, nitrogen is found in four different states of oxidation. One of these, nitrogen gas, is relatively inert and is not included in the total nitrogen measurement; however, it can be incorporated into the cycle through biological fixation by blue-green algae or can be lost from the biological cycle through the action of denitrifying microorganisms on nitrates, thus reducing the total nitrogen concentration. By analogy with the general development of the phosphorus models (Vollenweider, 1969), the change in total nitrogen concentration per unit time equals the rate of loading of nitrogen from external sources per unit area divided by the sum of mean depth, plus internal loading from the sediments plus the rate of nitrogen fixation minus losses through the outlet minus losses to the sediments minus denitrification losses. Some of the parameters in this equation are easily measured or estimated (total nitrogen concentration, areal nitrogen loading, lake mean depth, and hydraulic flushing rate), but the rest are very difficult if not impossible to measure. These factors (internal loading, sedimentation losses, nitrogen fixation, and denitrification) are grouped together as attenuation losses and are expressed as:

attentuation losses = α TN
were
α = attenuation coefficient, yr^{-1}

TN = the concentration of total nitrogen in the lake, mg. m^{23}

The differential equation is given as:

$dTN/dt = L/z - \quad TN - \quad TN$ (1)

where
t = time
L = annual nitrogen loading per unit of lake surface area,
$mg \cdot m^{-2} yr^{-1}$
z = mean depth of lake, m
ρ = hydraulic flushing rate, yr^{-1}

The steady-state solution is:

$TN = L/(Z(\alpha + \rho))$ (2)

This is the same as the solution for the total phosphorus model (Vollenweider, 1975) with the exception that the sedimentation coefficient has been replaced with an attenuation coefficient.

DATA BASE

The basic data were obtained from the results of the U.S. Environmental Protection Agency National Eutrophication Survey. Data were tabulated for all lakes on annual areal total nitrogen loading rates, median total nitrogen concentrations, lake mean depths, hydraulic flushing rates, chlorophyll *a* concentrations, total phosphorus concentrations, and total phosphorus areal loading rates. The median total nitrogen concentration was taken to represent the steady-state total nitrogen concentration, agreeing with Reckhow (1977) that the median would be less affected by extreme measurements. Nitrogen attenuation coefficients for each lake were estimated from the data by assuming steady state and rearranging the terms in Equation 1:

$$\alpha = L/(TNZ) - \rho$$

All the errors in estimating the total nitrogen concentration, areal loading, lake mean depth, and hydraulic flushing rate are incorporated into the attenuation coefficient. Negative values for this coefficient might indicate a lake that has a net production of nitrogen through nitrogen fixation, is not in steady state, or where the errors of estimation may result in a negative value.

The sample includes all the EPA-surveyed lakes with a complete set of data. In the first year of that survey, total nitrogen concentrations were not measured, thus reducing the size of the sample. The remaining 95 natural and 384 artificial lakes include a wide range of lake types with mean depths from 0.5 to 307 meters, total nitrogen concentrations from 125 to 7,185 mg m^{-3}, nitrogen loading from 1,500 to 14,900,000 mg $m^{-2}yr^{-1}$, and attenuation coefficients from -5 to 392 yr^{-1} (Table 1).

The lakes were randomly sorted into two data sets. One data set (model development) with 49 natural and 199 artificial lakes was used to develop the predictive models, and the other data set (model verification) with 46 natural and 185 artificial lakes was used to test the predictive abilities of the empirical models and establish confidence limits. Because the values of most parameters spanned several orders of magnitude and it was reasonable to assume that variances were proportional to means, all data were transformed to their natural logarithms before statistical analyses (unless stated otherwise.).

NITROGEN ATTENUATION COEFFICIENTS

Because the nitrogen attenuation coefficient cannot be directly measured, I investigated the possibility that it could be related to some other measurable variable. Correlations between the coefficient and several limnological variables are shown in Table 2. In general, stronger correlations were found for artificial than for natural lakes. The best correlations were obtained with various measures of water or nitrogen loading, with greater rates of input being associated with greater fractional losses of nitrogen from the lake water (Figure 1).

In addition, a nitrogen retention coefficent was calculated following the procedures that Dillon and Rigler (1974) used for phosphorus. The retention coefficient and its logarithms also were used in the same correlation matrix, but the resulting correlations were less strong than those found by using the attenuation coefficient. I also attempted to fit a nitrogen settling velocity following Chapra's (1975) work with phosphorus, but it also was less satisfactory.

The strongest correlations were found with the volumetric nitrogen loading, the areal nitrogen loading, and the hydraulic flushing rate: however, this does not prove a cause-and-effect relationship for any one variable. Indeed, these three variables are all intercorrelated (Table 3); any one of them could influence nitrogen attenuation, or there could be an important unmeasured variable that also is correlated with either nitrogen or water inputs.

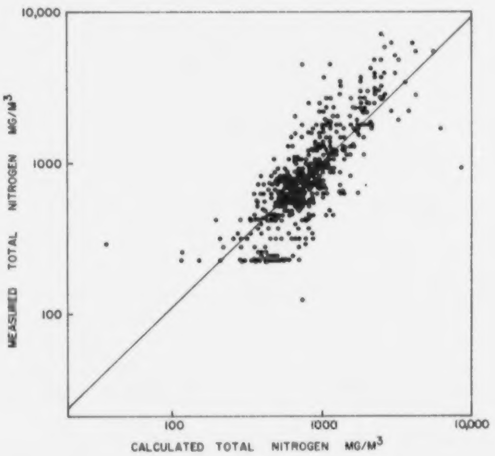

Figure 1. — Relationship between nitrogen attenuation coefficients and volumetric nitrogen loading for both natural and artificial lakes combined.

Table 1. — Mean values and related statistics for annual areal total nitrogen loading rates ($mg{\cdot}m^{-2}yr^{-1}$), total nitrogen concentrations ($mg{\cdot}m^{-3}$), mean depths (m), hydraulic flushing rates (yr^{-1}), and calculated attenuation coefficients (yr^{-1}), for 479 natural and artificial lakes included in this study.

Variable	Lake Type	No. in sample	Mean	Standard deviation	Range	
					Minimum	Maximum
Areal nitrogen loading (L)	natural	95	60894.0	172457.0	1500.0	14900000
	artificial	384	139092.0	635435.0	1700.0	11155000
Total nitrogen (TN)	natural	95	1441.0	1223.0	125.0	6040
	artificial	384	1027.0	974.0	220.0	7185
Mean depth (z)	natural	95	11.0	32.4	0.5	307
	artificial	384	9.2	8.4	0.6	59
Hydraulic flushing rate (ρ)	natural	95	4.9	8.5	0.002	45
	artificial	384	14.4	40.4	0.019	365
Attentuation coefficient (α)	natural	95	4.8	14.3	-8.3	130
	artificial	384	8.7	31.3	-50.0	392

Table 2. — Correlation coefficients (r) between various limnological parameters and attenuation coefficients. Logarithmic transformations were used. All coefficients significant at the 5% level except those marked NS not significant).

Parameter	Natural Lakes	Aritifical Lakes	Both Combined
Volumetric nitrogen loading	0.60	0.78	0.74
Areal nitrogen loading	0.67	0.75	0.74
Hydraulic flushing rate	0.63	0.74	0.72
Areal water loading	0.61	0.66	0.65
Mean depth	-0.12	-0.30	-0.25
Ratio total nitrogen to total phosphorus	-0.25	-0.18	-0.20
Chlorophyll *a*	0.00 NS	-0.06 NS	-0.05 NS
Total nitrogen concentration	-0.14 NS	-0.04 NS	-0.01 NS

Table 3. — Correlations between the logarithms of chlorophyll *a* (CHLA), areal phosphorus loading rate (LP), areal nitrogen loading rate (L), volumetric nitrogen loading rate (L/Z), total nitrogen (TN), total phosphorus (TP), ratio of total nitrogen to total phosphorus (TN/TP), hydraulic flushing rate (ρ), and the ratio of the areal nitrogen loading rate to the areal phosphorus loading rate (L/LP).

	CHLA	LP	L	L/Z	TN	TP	TN/TP	ρ	L/LP
CHLA	1.00	0.07	0.05	0.33	0.67	0.69	-0.22	0.12	-0.25
LP		1.00	0.63	0.56	0.22	0.34	-0.22	0.53	-0.20
L			1.00	0.86	0.32	0.22	0.04	0.83	0.06
L/Z				1.00	0.56	0.47	-0.05	0.89	-0.02
TN					1.00	0.66	0.15	0.27	-0.08
TP						1.00	-0.64	0.26	-0.52
TN/TP							1.00	-0.08	0.61
ρ								1.00	-0.02
L/LP									1.00

Regression equations were developed with the model-development data set for the relationships between the nitrogen attenuation coefficients and the volumetric nitrogen loading, areal nitrogen loading, and hydraulic flushing rates. These were determined for natural and artificial lakes both separately and combined (Table 4). The attenuation coefficients were then substituted back into Equation 2 to yield the various predictive models for total nitrogen.

MODEL VERIFICATION

I tested the abilities of these models to predict the measured total nitrogen concentrations of the lakes in the model-verification data set. Correlation coefficients were calculated between measured and calculated total nitrogen concentrations, and empirical 95 percent confidence limits were determined for the calculated total nitrogen concentrations of each model by calculating the standard deviation of the mean difference between the logarithms of the measured and calculated total nitrogen concentrations. Average errors and average percentage errors also were calculated from the untransformed calculated and measured total nitrogen values. These four measures of precision were used to evaluate the respective models.

Table 4. — Comparison of calculated and measured total nitrogen concentrations for the model-verification data set with use of models based on volumetric nitrogen loading (L/Z), areal nitrogen loading (L), and hydraulic flushing rate (ρ). Error estimates include the average error (AE), percentage error (PE), and 95% confidence limits as percentages of the calculated total nitrogen value (CL).

Model	Correlation coefficient r	Error estimates AE	PE	CL
Based on L/z				
natural lakes with				
$\ln \alpha = -0.345 + 0.505 \ln (L/z)$				
and artificial lakes with				
$\ln \alpha = -0.434 + 0.618 \ln (L/z)$	0.80	410	38	41-253
both with				
$\ln \alpha = -4.144 + 0.594 \ln (L/z)$	0.80	419	46	47-286
Based on L				
natural lakes with				
$\ln \alpha = -6.506 + 0.724 \ln L$ and				
artificial lakes with				
$\ln \alpha = -6.430 + 0.709 \ln L$	0.82	382	37	41-255
both with				
$\ln \alpha = -6.426 + 0.710 \ln L$	0.82	382	37	41-255
Based on ρ				
natural lakes with				
$\ln \alpha = -0.485 + 0.5861 \ln \rho$ and				
$\ln \alpha = -0.291 + 0.5821 \ln \rho$	0.76	513	59	36-325
both with				
$\ln \alpha = -0.367 + 0.5541 \ln \rho$	0.77	498	56	36-315

For the models based on volumetric loading, areal loading, and flushing rate, similar results (Table 4) were obtained whether separate equations were used for natural and artificial lakes or a single equation was used for both. This indicates that the same coefficients can be used in nitrogen models for both natural and artificial lakes. This contrasts with phosphorus models in which different coefficients for the respective lake types lead to a greater degree of precision (Canfield, 1979).

The best results were obtained for the models based on volumetric nitrogen loading (Figure 2) or areal nitrogen loading although the model based on flushing rate also gave acceptable results. It was originally thought that a simple nitrogen model would give poorer results than a phosphorus model because of the greater complexity of the nitrogen cycle. This was not found, for by comparison, the best available model for predicting total phosphorus (Canfield, 1979) had an average percentage error of 44 percent and 95 percent confidence limits of 31 to 288 percent when applied to a similar set of lakes. My best nitrogen models have a 37 percent percentage error and 95 percent confidence limits of 41 to 255 percent.

NITROGEN, PHOSPHORUS, AND CHLOROPHYLL *a*

Stronger correlations (Table 5) were found between total nitrogen and chlorophyll *a* in natural lakes than in artificial lakes. This agrees with similar findings by Canfield (1979) for the total phosphorus-chlorophyll *a* relationship and presumably results from a greater chance of light limitation in artificial lakes because of greater concentrations of inorganic particulate materials.

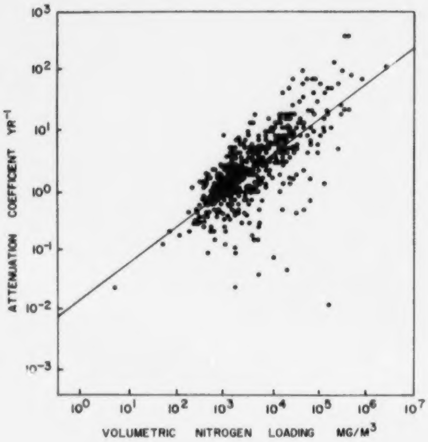

Figure 2. — Relationship between measured total nitrogen and total nitrogen calculated with separate regressions for natural and artificial lakes on the basis of volumetric nitrogen loading (Table 4). The best-fit linear regression line is shown.

Table 5. — Correlations (r) between logarithms of chlorophyll *a* and total nitrogen and total phosphorus for natural and artificial lakes.

	Natural lakes	Artificial lakes
Total phosphorus	0.84	0.59
Total nitrogen	0.81	0.59

The relatively high correlation (r = 0.81) between total nitrogen and chlorophyll *a* was unexpected, because most of the lakes were thought to be phosphorus-limited on the basis of the ratios of total nitrogen to total phosphorus (Table 6). Vallentyne (1974) reported that aquatic plants characteristically have ratios of nitrogen to phosphorus of about 7, considerably smaller than the average ratio of 23.7 in the sample lakes. Most likely, the high correlation is because of the fact that total nitrogen and total phosphorus concentrations in lakes are highly correlated with each other (Table 3), so that they both would be correlated with chlorophyll even though phosphorus may have been the limiting nutrient in most instances.

Other similarities were noted between the behavior of nitrogen and phosphorus in the lakes in this sample. In general, the lakes were sinks for both elements with similar loss rates for both as indicated by the finding that the average ratios of nitrogen to phosphorus within the lakes were not significantly different from the ratios in the inputs (Table 6). There were no large shifts in the ratio indicating differential losses or substantial effects of nitrogen fixation by blue-green algae.

The only major difference was found in those lakes (27 of 479) where negative nitrogen attenuation coefficients indicated that more nitrogen was being produced in the lake than was being lost. It may be significant that the average N:P ratio in the inputs to those 27 lakes (17.4) is significantly different from the ratio (24.5) in the other 452 lakes with positive attenuation coefficients, but the N:P ratios within the two groups (23.9 and 23.7, respectively) are not different. This could illustrate the proposal by Schindler (1977) that lakes with small ratios of N:P in their inputs will have enhanced rates of nitrogen fixation, with a subsequent elevation of the N:P ratio within the lakes themselves.

Table 6. — Frequency distributions of the ratios of total nitrogen (TN) to total phosphorus (TP) within the lakes in the sample and the ratios of the annual surface loading of total nitrogen (L) to the annual surface loading of total phosphorus (LP). The differences between the averages of the two ratios are not statistically significant.

Ratio TN:TP	% of lakes with a smaller ratio	Ratio L:LP	% of lakes with a smaller ratio
2	0.6	2	0.2
4	1.9	4	2.9
6	7.0	6	9.2
8	12.1	8	15.8
10	19.8	10	23.2
12	27.4	12	30.5
14	34.9	14	39.4
16	42.3	16	47.3
18	49.6	18	54.3
20	56.8	20	58.2
30	79.1	30	77.5
40	89.1	40	86.2
60	96.6	60	95.2
80	99.4	80	98.1
Average = 23.7			Average = 24.1
Std. Dev. = 20.9			Std. Dev. = 27.5

Lastly, the nitrogen attenuation coefficient and the analogous phosphorus sedimentation coefficient are both strongly correlated with the loading rates of the respective elements as well as with the water loading rates (Canfield, 1979), leading to similar forms for their respective prediction equations. The reasons for this are poorly understood. The strong affinity of phosphorus for particulate materials has been used as an explanation for its behavior (Canfield, 1979), but this does not seem likely for nitrogen. Clearly more work is needed to understand the factors controlling nitrogen concentrations in lakes.

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AN INCREMENTAL PHOSPHORUS LOADING CHANGE APPROACH FOR PREDICTION ERROR REDUCTION

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ABSTRACT

Lake quality management planning necessitates projecting the impact of proposed watershed activity and land use changes on lake quality. In most cases, the change is relatively small in comparison to the watershed characteristics that are expected to remain constant over the planning period. Prediction using a lake loading model is probably unnecessary for these unchanging land uses, since existing lake data represent the resultant water quality impact. In those situations, lake loading model prediction may be required only for the proposed watershed land use change(s). With sufficient representative lake quality data, the future projection reliability is improved when the model prediction is calculated for the change only. This is manifested in a reduction in the total projection error, which is a function of lake data variability (for unchanging land use), and model and loading error (for changing land use).

INTRODUCTION

Effective lake quality management planning necessitates the use of quantitative methods or models to relate relevant human activities and natural characteristics to lake water quality. Models, in turn, are of particular value to the planning process when reliability can be directly assessed. Reliability, or its converse, uncertainty, serves three vital functions in planning studies:

1. Reliability represents an estimate of the value of information. If the reliability associated with a prediction is low, the prediction is uncertain and imprecise, and the predictive information is not particularly valuable. Alternatively, if the reliability of a prediction is high, the prediction is precise, and the predictive information can be valuable.

2. Important factors that are poorly characterized (i.e., have high uncertainty) may be identified when reliability is assessed. The analysis of uncertainty, or error, helps model developers and model users in a sensitivity analysis exercise. Specifically, estimation of errors allows the analyst to identify those characteristics that have significant error and have a significant effect on the prediction. The analyst then realizes that in order to obtain a precise prediction, these uncertain, prediction-sensitive terms must be better defined.

3. Discrimination among control strategies may be explicitly evaluated with an assessment of reliability. Without uncertainty analysis, one is given the impression that prediction differences of one microgram per liter or less are significant and indicate a well-defined ordering of quality states. With uncertainty analysis, the prediction interval, or confidence interval defined by the prediction error, identifies a region in which land use strategies may be predictively indistinguishable. The error analysis allows the planner to determine when land use strategy impacts can be predictively distinguished, given the error associated with model applications.

Several phosphorus lake models have been proposed recently that incorporate a procedure for estimating prediction uncertainty (Chapra and Reckhow, 1979; Reckhow, 1979a, b; Reckhow and Simpson, 1980; Reckhow and Chapra, 1980; Reckhow, et al. 1980). These approaches represent an improvement over the purely deterministic analyses presented in older literature, since the error estimate is a measure of prediction information value.

However, there are two problems or shortcomings with the existing error analysis methods. Errors arise in model applications because of error in the model, the model parameters, and the model variables. In a more fundamental sense, one may also say that the errors are caused by natural variability, inadequate sampling design, measurement error and bias, and model specification error. When the data set variables used to construct a model contain error, then this error is transmitted to the model error term for the fitted model. Since virtually all limnological statistics contain error as a result of the aforementioned causes, then lake models developed from these data contain error associated with the data error. This means that the model standard error term includes an error component associated with errors in the model variables. This is an unwanted component, yet it is unavoidably there given present knowledge and data.

Models can be employed in a descriptive or a predictive mode. When used descriptively, models may be used to relate observed inputs (the independent variables) to observed outputs (the dependent variable). In a descriptive application, when all variables are directly measured in the same manner as the variables in the model development data set were measured, then there is no need to add additional application lake variable error. This is because the appropriate variable error is already contained in the model error term. However, when the model is used in a predictive mode, the dependent variables generally cannot be measured (because the predictive nature implies conditions not yet physically realized). Predictive applications of a model require that the analyst extrapolate variable values from other points in time and/or space. This extrapolation process introduces error beyond that already contained in the model error term. Thus, predictive use of a model should be accompanied by an error analysis that includes variable error. This errors-in-variables analysis must be undertaken thoughtfully, however, to avoid "double counting" errors (due to the errors-in-variables term already contained in the model standard error term). The first problem of existing error analysis methods, therefore, is that their application may lead to error double counting.

The second shortcoming associated with existing methods is of greater importance, given the fact that with care, double counting can probably be kept at an acceptably low level. The second problem relates to the magnitude of the error term. The input-output empirical phosphorus lake models of concern here are developed from cross-sectional analyses. The models are simple, our knowledge of limnology is limited, and all phosphorus-settling processes are aggregated into one empirically-determined model parameter. As a result, the model error term is large, since it represents cross-sectional variability, measurement and sampling error, and model specification error. In addition, errors in the model variables, particularly in phosphorus loading, can be substantial for certain applications. The combined effect of these error terms is a large prediction error using existing error analysis methods. The magnitude of ths error term, and the associated prediction intervals, is such that the analyst is often unable to find "statistically significant differences" among competing lake management options.

A PROPOSED ERROR ANALYSIS METHODOLOGY

An alternative error analysis methodology will substantially reduce prediction error over existing error analysis techniques for most applications. This procedure exploits two features that are common to many lake quality management planning situations:

1. For most projected planning scenarios, the land use area expected to change is small in comparison to the land use area that is expected to remain constant during the planning period. Stated another way, the impact of the change is generally small in comparison to the impact of the existing land uses.

2. Existing phosphorus lake data reflect the impact of present land use conditions. Furthermore, the variability in these data represent the variability in impact response. These data could already be in existence or they could be acquired upon initiation of this modeling program.

To see how these features can lead to prediction error reduction, consider a planning scenario in which no change is projected so that existing land use and future land use are equivalent. In that case, two methods may be used to predict future lake phosphorus concentration (ignoring temporal variability for the moment):

1. A phosphorus lake model may be applied to relate land use to phosphorus concentration through literature export coefficients. This standard procedure is accompanied by a high prediction error.

2. Existing phosphorus lake data may be used to describe future lake quality under unchanging watershed conditions. Here the error term is a function of the standard error of the estimate for the data and of the representativeness of the data.

In virtually all cases, with even a modest amount of phosphorus lake data, the error for the second method will be considerably smaller than the error for the first method, given the size of the phosphorus loading and model error terms.

If this scenario is modified slightly to a situation common in lake quality management planning, the new error analysis methodology may be outlined. Consider a planning scenario in which a relatively small land use change is projected. The new modeling and error analysis methodology stipulates that the analyst use:

1. Existing lake data to evaluate the impact of unchanging land uses, and

2. The model to evaluate the impact of the land use change and the impact of hydrologic variability.

Existing error analysis methods do not permit the analyst to distinguish between land uses that are projected to change and land uses that are to remain constant. This means that the impacts of all watershed land uses on lake phosphorus concentration are evaluated through the model. Since the impact of unchanging land uses is manifested in recent lake phosphorus concentration data, information (the lake phosphorus concentration data) is wasted and high prediction errors result.

To indicate the magnitude of the error reduction associated with the procedure outlined herein, consider the following model (Reckhow, 1979b):

$$P = \frac{L}{11.6 + 1.2q_s} \qquad \text{Eq. (1)}$$

where:

P = lake phosphorus concentration (mg/l)
L = annual areal phosphorus loading (g/m^2-yr)
q_s = annual areal water loading (m/yr)

The model standard error is .128 in logarithmically-transformed concentration units. This translates to about a ± 30 percent prediction error when the antilog is determined for a particular concentration. The difference between the existing and proposed error analysis methodologies may best be stated through hypothetical comparisons.

1. **With the existing error analysis procedures**, the model is used to predict the impact from all land uses. The model error alone (to which errors in variables must eventually be added) is approximately ±30 percent. For oligotrophic lakes, this model error term is relatively small. However, planning frequently occurs on lakes with phosphorus concentrations ranging from

.020 mg/l to .060 mg/l. Plus or minus 30 percent error would amount to ± .006 mg/l to ± .018 mg/l l for these concentrations. This is a substantial error term, and it may both discourage the planner from using error analysis and obscure the differences among management strategy impacts.

2. With the error analysis procedure proposed herein, the model is used to predict the impact for the changing land uses only. The analyst must use the model to evaluate the impact for both the old and the new land used. Most projected changes in land use have a relatively minor impact on lake phosphorus concentration in comparison to the impact from all watershed land uses. For comparison purposes assume that a land use change from forest to agriculture is to occur in a watershed. Assume that model predictions indicate that this forested land contributed 2 percent of the total phosphorus loading to the lake and that the new agricultural use is expected to contribute about 10 percent. Since the model must be used to evaluate the impact of both old and new changing land uses, the result is a 12 percent (10 + 2) loading change to be evaluated using the model. For the range of lake phosphorus concentrations of .020 mg/l to .060 mg/l, and a model error of ± 30 percent the model prediction error term is .00072 mg/l to .00216 mg/l. If a reasonable amount of lake sampling for phosphorus concentration has occurred (under a good sampling design), then the impact of unchanging land uses may be evaluated objectively. Even for modest amounts of data, the standard error will usually be small.

For example, Reckhow (1979c) evaluated phosphorus data variability in a cross-sectional study and found that the interquartile range is equivalent to about half the median phosphorus concentration. If it is assumed that the interquartile range is approximately twice the standard deviation,and the mean and median are equivalent, then the standard deviation is about one-fourth of the mean. For the phosphorus concentration range of .020 mg/l to .060 mg/l, the estimated standard deviation is about .005 mg/l to .015 mg/l. With a relatively small data set of perhaps 20 to 30 phosphorus concentration measurements, the standard error of the estimate is ($1/\sqrt{n}$ times the standard deviation) .00091 mg/l to .0034 mg/l. Combining this error term with the model prediction error term for changing land uses (square the error terms, add, and calculate the square root), the error ranges from .0012 mg/l to .0040 mg/l.

This comparison does not include all error terms for either methodology (see Reckhow, 1980, for an example with more detail), but the major error terms are calculated. Note that the proposed methodology reduces prediction error by 75 to 80 percent in the hypothetical example. The analyst must realize, however, that this error reduction associated with the new methodology is contingent on the magnitude of the land use change that must be evaluated using the model. Obviously, as the magnitude of the projected impact increases, the advantage of the proposed procedure diminishes.

The hypothetical example comparing error analysis procedures includes three of the four basic terms for the proposed methodology. The four error terms, and their interpretations in modeling applications, are:

1. Uncertainty in the assessment of current lake phosphorus concentration. If adequate data exist, this error term may be represented by the mean square error of the data. Data "adequacy" should be determined by whether existing data are representative on a spatial and temporal (within and across years) basis. In situations with inadequate data, this error term may be estimated through regression analysis with more comprehensive data sets on correlated variables (e.g., Secchi disk transparency) or through subjective determination.

2. Uncertainty in the hydrology variable, q_s. Cross-sectional error in q_s already exists in the model standard error for the reason identified earlier. This q_s-component of model error has unknown magnitude and may be sufficient for lakes with low inflow-outflow variability. Further, when several years of in-lake phosphorus concentration data exist (described in error component number 1), these data already exhibit the effect of q_s-variability, making q_s-error analysis unnecessary. Therefore, this additional error term may be considered optional. In cases with substantial variability in year-to-year values of the hydrology variable (q_s), and limited in-lake phosphorus data, a q -error term should be included, propagated through the model (using first order analysis: see Benjamin and Cornell, 1970). This error term should represent the year-to-year variability and the estimation or measurement error associated with the determination of q_s.

3. Uncertainty in the prediction of the impact of the projected new land use on lake water quality. This term is estimated using the phosphorus lake model and a procedure like that presented in Reckhow, et al. (1980). This error component includes model error and error in the estimate of phosphorus loading for the projected land use change. Note that for minor land use changes (relative to the entire watershed) the impact of this error term is small (despite the inclusion of model error) because the fractional phosphorus loading addition is small.

4. Uncertainty in the prediction of the impact of the existing land use in the area to undergo change. To properly assess the anticipated change, the analyst must determine the impact of both the old and the new land uses using the modeling/error analysis procedure. These calculations are undertaken in the same manner as are the calculations for error term described in Number 3. Note that here, too, the error is small when the fractional phosphorus loading subtraction is small.

In summary, the two error analysis methodologies may be compared with the aid of Figure 1. At the top of the figure, the traditional method is undertaken by estimating the phosphorus loading, and the loading estimation error, for all land uses in the lake watershed. The phosphorus loading and the loading error are propagated through the model for the calculation of the predicted lake phosphorus concentration. Prediction error for this procedure is determined by the loading error and the model error. Since the model error term is proportional to the phosphorus loading magnitude propagated through the model, and since all phosphorus loading is propagated through the model under the traditional procedure, the model error term is large. As a result, the total prediction error for the traditional procedure is often ± 30 to ± 40 percent

The new procedure often leads to a prediction error reduction because it is not required that the model (and model error) be used to predict all land use impacts. The watershed may be divided into land uses that are expected to remain constant over a planning period and land uses that are expected to change. Similarly the average phosphorus concentration in a lake may be divided into a fraction contributed by unchanging land use and a fraction contributed by land use that is expected to undergo change. For the new error analysis procedure, existing phosphorus lake data represent the impact of all existing land uses. The variability in these data reflect estimation uncertainty. Since no predictive model was required to assess this impact, the uncertainty term is often small. Added to this uncertainty is the prediction error associated with the determination of the impact of all changing land uses calculated using the model. However, since a fraction (often a sizable fraction) of the land use impacts is assessed without the model, total prediction error for the new procedure is generally much lower than it is for the old procedure.

ISSUES FOR CONSIDERATION

An effort has been made in this brief paper to stress a conceptual discussion of error analysis, forsaking at present applications and the mechanics of calculations. Continuing along this line of approach, some issues that were alluded to warrant yet further consideration. These issues are identified here in the hope they will stimulate additional analysis of this topic.

1. What is, or should be, the meaning of the lake phosphorus measurements error term? It is intended to represent the impact of unchanging land use on lake water quality.

a. Can we determine whether the lake is in steady state relative to watershed land uses?

b. It has been indicated that the lake phosphorus measurements should represent spatial and temporal variability. How does the need for temporal variability representation conflict with the need for "steady state" and the likelihood that most lakes undergo continuous small land changes?

c. Can "adequate" sampling design be defined objectively?

2. To what extent is time series variability represented in the lake phosphorus measurements and to what extent must it be included in the q -error term?

3. Time series data for q_s-variability could be extrapolated from other similar watersheds or perhaps from precipitation data. In those situations, an additional error term should be included, representing possible bias associated with the use of extrapolated data.

4. The analyst should be aware of the difference between the standard deviation and the standard error of the estimate. The standard deviation is a measure of the variability in a set of data. The standard error of the estimate, which may often be calculated from the standard deviation by dividing by $\sqrt{n}$, reflects the error in a statistic. The error analysis yields a standard error of the estimate that represents the error in the prediction; it does not directly reflect the variability to be expected for (in this case) lake phosphorus concentration.

5. The error propagation equation (Benjamin and Cornell, 1970) is to be used to calculate the impact of errors in the variables and errors in the parameters on the total prediction uncertainty. One term in the error propagation equation represents the error contribution associated with variable (or parameter) correlation. Should this term be computed for the correlation between the old, changing land use phosphorus loading and: (a) the new land use phosphorus loading, and/or (b) q_s? This question is largely of a conceptual nature, since the impact on total prediction uncertainty in either case is undoubtedly quite small. Nevertheless, it illustrates the type of conceptual problem that must be considered as error analysis methodologies are proposed and refined.

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APPLICATION OF PHOSPHORUS LOADING MODELS TO RIVER-RUN LAKES AND OTHER INCOMPLETELY MIXED SYSTEMS

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ABSTRACT

Theoretical calculations are used to illustrate how river-run reservoirs tend to retain a larger fraction of their phosphorus loading than completely mixed lakes because of the effect of incomplete mixing or the sedimentation process. Empirical models are used to demonstrate the correlation between flushing characteristics and sedimentation. Enhanced settling is also ascribed to the higher proportion of solid-associated phosphorus in the loadings of incompletely mixed systems. The importance of solids to lake phosphorus budgets is demonstrated with a nutrient/phytoplankton model for a river-run lake.

INTRODUCTION

The phosphorus loading concept provides a variety of mathematical and graphical models to predict trophic state as a function of simple expressions of a lake's morphometry, hydrology, and loading. Because they can be used to make inexpensive, order-of-magnitude estimates of water quality, these models have been widely applied for lake management. However, as with any mathematical idealization, there is a residual variability which these models do not explain.

While the variability of phosphorus loading models results from a variety of factors, it can be divided generally into two components (Chapra, 1980). The first, called "perceptual error," relates to our ability to perceive the actual state of an individual lake. Thus, perceptual error is caused by factors such as measurement errors and year-to-year meteorological variations that cause a lake to vary from its most likely condition. Hypothetically, if enough of the proper measurements are taken, the perceptual error (or the mean) would approach zero and we would obtain an accurate estimate of the "true" state of the lake.

The second component of the variability, called "lake uniqueness," relates to the fact that, even if the perceptual error is reduced to zero, an individual lake will still differ from model predictions because of biological, chemical, and physical factors not accounted for by simple phosphorus loading relationships. A case in point is Lake Washington where, even though phosphorus levels remained constant, its water clarity has recently increased because of changes in its zooplankton assemblage (Edmondson, 1978). In fact, Shapiro (1979) has suggested that biological factors not accounted for by simple models could represent viable control options for lake rehabilitation.

Whereas a strong case has been made for biological factors, less has been done to elucidate physical mechanisms that bear on phosphorus loading predictions (Chapra, 1979). The present paper is devoted to one of the more important physical aspects of lake uniqueness — incomplete horizontal mixing.

The theoretical basis of most phosphorus loading relationships developed to date is the completely mixed model (Figure 1). In this idealization, it is assumed that phosphorus inputs are instantaneously dispersed throughout the lake's volume so that the concentration in the water is homogeneous. While this is an excellent model for many lakes, there are a variety of systems where it does not apply (Figure 2). For example, river-run lakes and reservoirs typically exhibit strong horizontal gradients near river mouths and sewage outfalls that would not be accounted for by a well-mixed approach. Additionally, incomplete mixing is

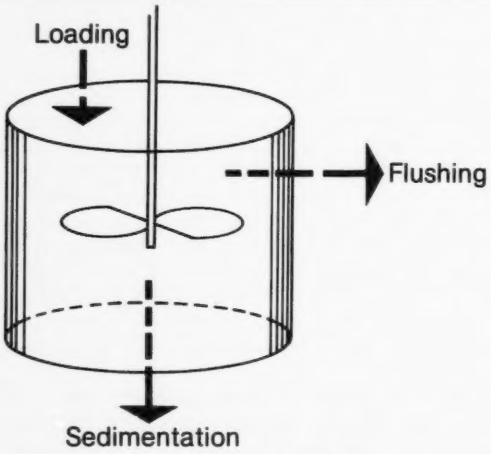

Figure 1. — A completely mixed model showing the majoı mechanisms governing the level of total phosphorus in a lake.

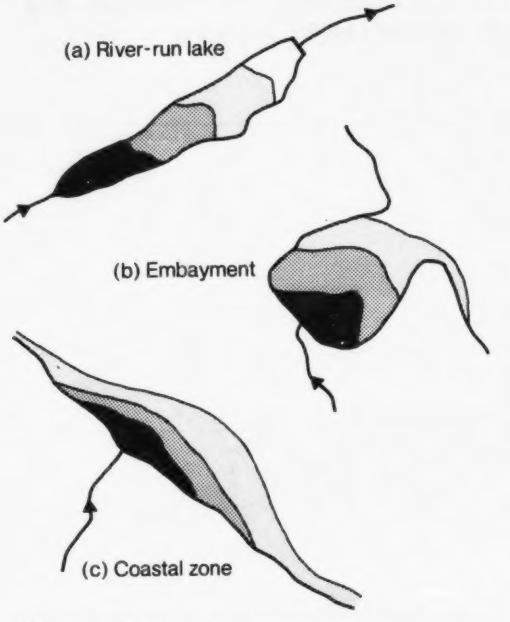

Figure 2. — Overhead views of some incompletely mixed systems. The darkshaded areas represent heightened phosphorus levels near river mouths and the arrows designate the direction of flow.

relevant to the modeling of lake sub-areas such as embayments and the littoral zone where human influence, use, and perception of the water body are intense. Thus, the application of completely mixed models to such systems, neglects, or averages out, the inhomogeneities that represent critical aspects of their water quality

Of equal importance is the fact that neglecting these gradients and their underlying mechanisms may lead to faulty predictions when applied to incompletely mixed systems. In the present paper, this is illustrated by comparing the prediction of a well-mixed model with the prototype incompletely mixed system — the river-run lake or reservoir. These elongated lakes are ideal for such a contrast since they exhibit most of the characteristics of other incompletely mixed systems yet have a simple one-dimensional transport regime that allows a clear perception of the processes underlying their dynamics. The basic conclusion of the comparison is that incompletely mixed systems are more efficient sedimentation basins than well-mixed lakes. The interrelationship of settling and lake hydraulics is also demonstrated by a theoretical analysis of some empirical phosphorus loading models. Finally, the importance of solids to the dynamics of incompletely mixed systems is demonstrated by a nutrient-food chain-suspended solids model for a river-run lake.

THEORETICAL COMPARISON OF COMPLETELY MIXED AND RIVER-RUN BUDGET MODELS

Input-output or budget models predict a lake's contaminant level by determining fluxes of the substance across the system's boundaries. The input of phosphorus consists of loadings such as sewage effluents and tributary discharges that enter the lake at its periphery or atmospheric loadings that enter through its surface. In general, two major processes characterize phosphorus losses. The first, *sedimentation,*represents the net amount of phosphorus incorporated into the lake's bottom along with settling particulate matter. The second, *flushing*, represents the loss of phosphorus carried by water flowing through the lake's outlet.

As depicted in Figure 1, the assumption of complete mixing allows these processes to be modeled in a very simple fashion. For example, since the point of entry of the inputs is irrelevant, a single term can be used to represent the total loading. In a similar fashion, sedimentation and flushing can be represented by simple formulations. For systems where mixing is not complete, however, adequate characterization of in-lake water motion or transport is required.

As will be shown, the more complex transport regime in turn has an impact on the magnitude and structure of the flushing and sedimentation processes and requires that the location of inputs be specified. Before demonstrating the importance of these factors for a river-run lake, however, the completely mixed model will be reviewed briefly as a point of reference for the subsequent discussion.

The Completely Mixed Model

A phosphorus budget model for a well-mixed lake can be expressed mathematically as (Vollenweider, 1969; Chapra, 1975)

$$V\frac{dp}{dt} = W - Qp - vA_sp \qquad (1)$$

(accumulation) = (inputs) - (flushing) - (sedimentation) where V is lake volume ($10^6 m^3$), p is its phosphorus concentration (mg m^{-3}), t is time (yr), W is the rate of mass input of phosphorus (kg yr^{-1}), Q is the rate of water flow through the lake's outlet (10^6 m^3 yr^{-1}), is the apparent settling velocity of total phosphorus (m yr^{-1}) and A_s is the lake's surface area (10^6 m^2).

At steady state (i.e., dp/dt = 0), Eq. 1 can be solved for

$$p = p_{out} = p_{in}\left(\frac{1}{1 + v/q_s}\right) \qquad (2)$$

where p_{out} is the total phosphorus concentration of the outlet (mg m^{-3}), P_{in} is the concentration of the inputs (mg m^{-3}) = W/Q, and q_s is the areal water loading (m yr^{-1}) = Q/A_s. Thus, Eq. 2 is a relationship that can be used to calculate in-lake phosphorus concentration as a function of loading and parameters related to the lake's flushing and sedimentation characteristics. Note that since the lake is well-mixed, the concentration of outflowing water is equivalent to that at mid-lake. In the following section, this equivalence is contrasted with systems when in-lake concentrations are heterogeneous.

The River-Run Model

In contrast to the disorganized or turbulent flow regime of the well-mixed lake, a river has a well-organized, unidirectional flow as depicted in Figure 3a. For the ideal case where no longitudinal mixing is present, the river flow would not change the identity of the substance being transported. Thus, as in Figure 3b, a conservative dye (i.e., one which does not react or settle), would merely move downstream along with the water flow. In the engineering lexicon, such systems are called *plug-flow reactors*. For the case where a substance settles at a first order rate as it flows, it can be shown (Reckhow and Chapra, in press) that the steady state concentration downstream from a constant point source can be calculated as

$$p = p_{in} \exp[-(vw/Q)x] \qquad (3)$$

where w is the width of the river (km) and x is the distance downstream from the waste source (km). As in Figure 3c, note that while the substance maintains its longitudinal identity, it gradually diminishes in concentration because of settling losses. However, in comparison to the well-mixed model, the concentration along the longitudinal axis of the river is not homogeneous. Thus, the outlet concentration differs from the mid-lake value. If the "outlet" for the river is defined as being at distance L downstream from the waste source, Eq. 3 can be used to calculate the concentration at that point as

$$p_{out} = P_{in} \exp(-v/q_s) \qquad (4)$$

Between the idealizations of complete mixing and plug flow are those lakes where both advection and turbulent mixing are important. Such river-run lakes, as depicted in Figure 4a, are typically long and narrow with a major tributary at one end and an outlet at the other. A key feature of such sytems is that advective water movement due to inflow and outflow is large enough to have a comparable effect on material transport as that caused by turbulent mixing due to winds and density difference. For such systems a plug of conservative dye introduced at the head end of the lake would move downstream along with the net water flow but would also spread out due to turbulent mixing as in Figure 4b. A steady state solution for such systems comparable to Eqs. 2 and 3 can be obtained (Reckhow and Chapra, in press) and is depicted graphically in Figure 4c. Note that for high levels of turbulent mixing, the solution becomes equivalent to the completely mixed model and for zero turbulence converges on the plug-flow model.

This exercise leads to the general conclusion that, all other things equal, a river-run reservoir is a more efficient settling basin than a completely mixed lake. This can be seen by observing that the outlet concentration (i.e., at x = L) for the well-mixed system is higher than for the river-run lakes. Thus, the amount of phosphorus retained by the latter would be higher. This is a necessary consequence of the direct, linear proportionality with concentration that is used to characterize sedimentation for both systems. In the well-mixed lake, sedimentation is uniform throughout the reactor since concentrations are homogeneous. In contrast, for the river-run lake settling is greater near the inlet where concentrations are high. These losses are proportionately more efficient than the reduced sediment losses near the outlet and the effect is that the net removal is higher than for the well-mixed case.

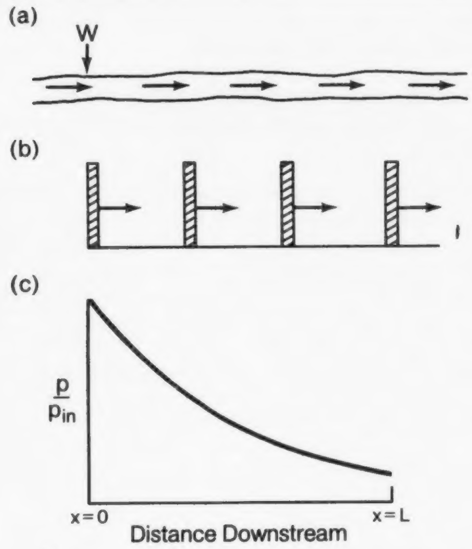

Figure 3. — A river with waste source at x =, (a) overhead view, (b) movement of a plug of conservative dye downstream, and (c) steady state profile of concentration normalized to concentration @ x = 0 for a substance that settles at a first order rate.

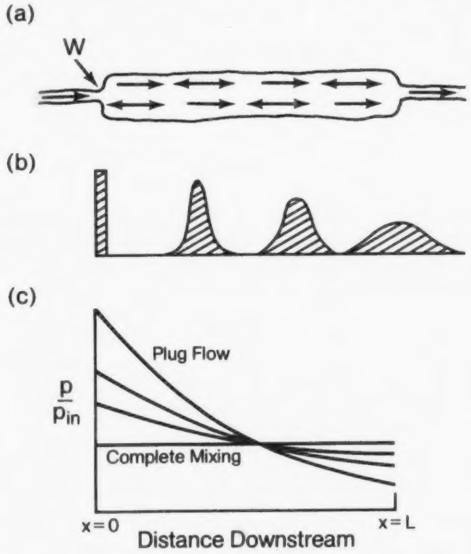

Figure 4. — A river-run lake with waste source at x = 0, (a) overhead view, (b) movement of a plug of conservative dye through the lake, and (c) steady state profiles of concentration normalized to inflow concentration for a substance that settles at a first order rate. The different profiles are for varying degrees of turbulent mixing.

This exercise provides a theoretical basis for the importance of sedimentation in incompletely mixed systems. While it has been limited to river-run lakes, similar processes would be evident in embayments and near-shore areas where loadings enter at the system's periphery and concentration gradients are pronounced. Before pursuing this subject with a more realistic model, the following section presents some empirical evidence along the same lines.

EMPIRICAL EVIDENCE LINKING FLUSHING AND SEDIMENTATION IN LAKES

A number of phosphorus loading models have been developed by fitting equations to budget data from sets of lakes. While some of these models have a semi-theoretical basis, many are strictly empirical and it is often difficult to determine what they imply regarding the cause and effect relationships underlying lake dynamics. For example, several models have been developed to predict the fraction of a lake's loading that does not exit via the outlet. The first of these retention models was that of Kirchner and Dillon (1975)

$$R_P = 0.426 \exp(-0.271\ q_s) + 0.574 \exp(-0.00949\ q_s) \quad (5)$$

where R_p is the retention coefficient. Eq. 5 seems to suggest that retention is solely dependent on the lake's hydraulic characteristics. However, as discussed previously, R_p is also a function of its sedimentation rate. The use of a single coefficient to define the combined magnitude of these processes can, therefore, obscure their individual effects. In contrast, theoretical models provide a means for keeping the mechanisms separate. For example, a theoretical retention coefficient for a completely mixed lake can be derived (Chapra, 1975) by rearranging Eq. 1 at steady state to yield

$$R_p = \frac{v}{v + q_s} \quad (6)$$

Note that in contrast to Eq. 5, the theoretically derived coefficient has separate terms for the flushing (q_s) and sedimentation (v) effects. Algebraically, Eq. 6 can be rearranged to yield

$$v = \frac{R_p q_s}{1 - R_p} \quad (7)$$

Eq. 5 can then be substituted into Eq. 7 to give (8)

$$v_{KD} = \frac{[0.426 \exp(-0271\ q_s) + 0.574 \exp(-0.00949\ q_s)]q_s}{1 - 0.426 \exp(-0.271\ q_s) - 0.574 \exp(-0.00949\ q_s)}$$

where v_{KD} is the apparent settling velocity of the Kirchner-Dillon model.

In essence, the above operation has separated the flushing and sedimentation effects that were confounded in the original model. The validity of the derivation depends on the assumption that the flushing mechanism for the lakes used to fit the Kirchner-Dillon model obeys the simple theoretical relationship in Eq. 1, i.e., that the outflow of mass equals the product of flow and concentration. If this is true, the manipulation has essentially removed the flushing effect from the retention relationship so that the residual (Eq. 8) is solely representative of the sedimentation process.

This operation can also be performed on other phosphorus loading models (Reckhow and Chapra, in press) with the results displayed in Figure 5. The surprising result is that even after the correction for flushing is made, it appears that sedimentation is still related to q_s.

Reasons for the positive correlation of v with q_s are presently a matter of speculation. A possible explanation is that the assumption of complete mixing and ideal flushing is being systematically violated. Chapra (1975) speculated that lakes with high values of q_s could be governed by different mechanisms than lakes with low q_s. Reckhow (1977) has suggested that lakes with $q_s > 50$ m/yr typically receive 90 percent or more of their inflow from one tributary. In other words, they may actually represent a distinctive class in that they are frequently just widened sections of rivers (i.e., river-run lakes). As shown previously, such lakes have different sedimentation characteristics than completely mixed water bodies; this might account in part for the effect. Further, lakes whose inputs come primarily from a major tributary (rather than from treatment plants) may have more of their phosphorus loading associated with eroded particulate matter that would settle quickly upon entering the lake. Thus, such lakes would have a higher apparent settling velocity.

While the foregoing is somewhat speculative it suggests the importance of sedimentation mechanisms for incompletely mixed systems. For this reason, the following section presents some theoretical computations to assess the impact of solids' dynamics on a river-run lake.

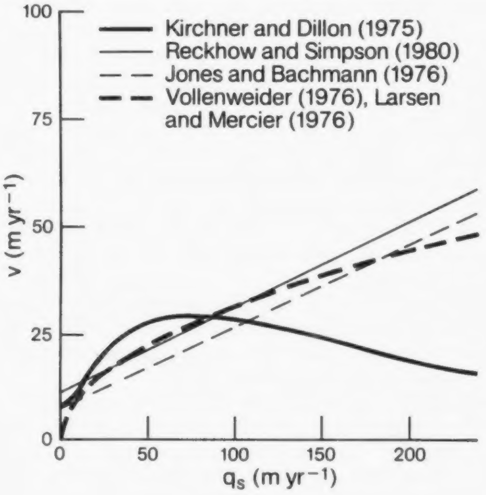

Figure 5. — Plot of apparent settling velocity (m yr^{-1}) versus a real water load (m $^{-1}$) for some commonly used phosphorus loading models. (Note: the Vollenweider and Jones and Bachmann models are also dependent on depth with shallower lakes having smaller settling velocities than deeper lakes.)

THE EFFECT OF SOLIDS ON PHOSPHORUS DYNAMICS OF A RIVER- RUN LAKE

One objection to the use of total phosphorus loading as a determinant of lake eutrophication is that a portion of such input is associated with particulate matter that settles rapidly upon entering a lake and, thus, never influences mid-lake quality (Schaffner and Oglesby, 1978). As suggested by the previous analysis, theoretical and empirical evidence suggests that such a process could be especially important for incompletely mixed systems. However,the models developed in the previous sections of this paper are unsuitable for a more detailed analysis of this phenomenon since they use a single variable, total P, to define the nutrient.

Therefore, a more detailed model that differentiates between various forms of phosphorus has been developed. As illustrated in Figure 6, the model consists of two particulate and two dissolved fractions. Inorganic particulate P (i.e., associated with inorganic particles such as fine-grained suspended sediments) adsorbs and desorbs dissolved inorganic P via equilibrium relationships. Phytoplankton P, on the other hand, is modeled kinetically and takes up dissolved inorganic P via a Michaelis-Menten relationship and releases phosphorus to the dissolved organic pool via a first order reaction. Dissolved organic P is, in turn, recycled to the dissolved inorganic pool by a first order reaction. In addition, the particulate fractions are lost via sedimentation with the inorganic matter settling at a somewhat higher rate. Details of the model's structure are described elsewhere (Reckhow and Chapra, in press).

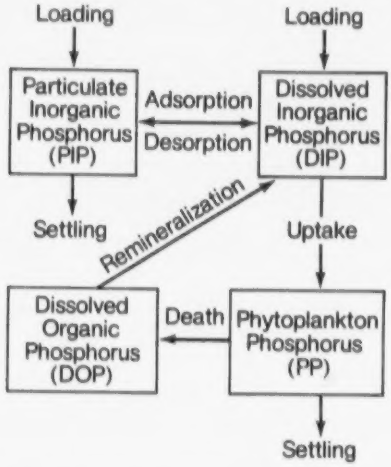
Loading
Loading
Particulate Inorganic Phosphorus (PIP)
Adsorption
Desorption
Dissolved Inorganic Phosphorus (DIP)
Settling
Remineralization
Uptake
Dissolved Organic Phosphorus (DOP)
Death
Phytoplankton Phosphorus (PP)
Settling

Figure 6. — Schematic of multi-species phosporus model. One-way arrows designate mass transfer mechanisms that are modeled kinetically. The two-way arrow specifies that sorption is treated as an equilibrium reaction (i.e., it is modeled using a partition coefficient.

The model was applied to a river-run lake with loadings of solids and phosphorus entering at the head end. The results of the simulation are shown in Figure 7. Note that the inorganic particles are at a high level at the beginning of the lake but eventually are removed from the water column (along with considerable quantities of adsorbed phosphorus) via sedimentation. In addition, the solids affect productivity by light attenuation with the result that phytoplankton growth is suppressed for most of the lake. Thus, while inorganic particles transport phosphorus into the system, their tendency to diminish water clarity and to remove P from the water via sedimentation tends to inhibit productivity.

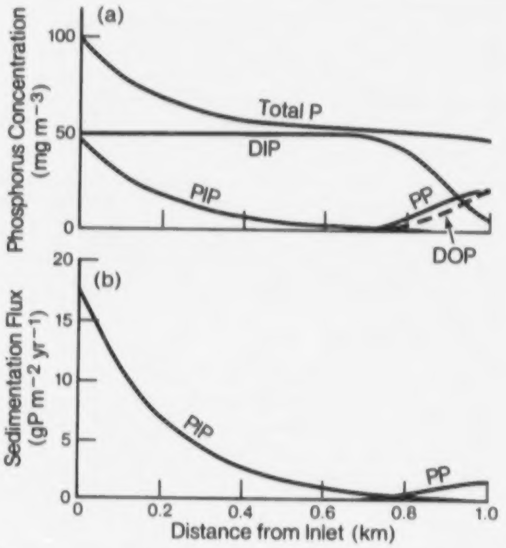

Figure 7. — Plots of (a) phosphorus concentration (mg m^{-3}) and (b) phosphorus sedimentation flux (g P m / yr^{-1}) versus distance downstream from the inlet of a river-run lake.

The importance of these factors to remedial control measures is demonstrated in Figure 8 where the effects of two alternative phosphorus abatement strategies are simulated. In the first, phosphorus loading is controlled by lowering the dissolved inorganic fraction with no effect on the incoming solids as might be the case for point source treatment. The result (Figure 8a) is that the phytoplankton levels are decreased in proportion to the load reduction. Figure 8b, on the other hand, shows the results if the solids loading is removed along with the phosphorus as might be the case if land runoff control were implemented. In this simulation, the peak phytoplankton level is higher than in Figure 8a because less P is removed from the water by sedimentation of inorganic particles. Additionally, the extent of phytoplankton growth increases to encompass most of the lake because of the absence of light attenuation by the inorganic solids. Thus, from the standpoint of productivity the latter control measure results in a more highly degraded lake than before treatment. Although this computation is a somewhat simple representation of a complex system, it serves to illustrate the importance of solids to the dynamics of incompletely mixed lakes.

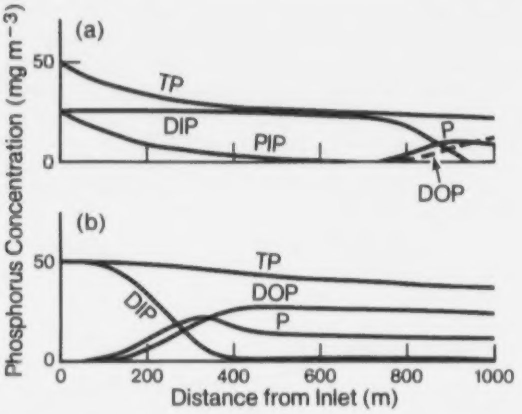

Figure 8. — Plots of phosphorus concentration versus distance downstream from the inlet of a river-run lake where (a) the phosphorus laoding is reduced by 50 percent with no solids control and (b) the phosphorus loading is reduced by 50 percent and all particulate inorganic solids are removed.

DISCUSSION

A specific objective of the foregoing analyses has been to demonstrate how the modeling of the dynamics of incompletely mixed systems is inextricably tied to the fate of solids. It should be noted, however, that this conclusion is also relevant to well-mixed lakes. As stated previously, Schaffner and Oglesby (1978) have suggested that dividing phosphorus loadings into available and non-available (i.e., rapidly settling) fractions could improve predictive models for well-mixed lakes. In addition, while the present paper dwells on horizontal features of lake physics, solids can also have an effect on vertical aspects.

Aside from thermal stratification, the primary vertical process influencing phosphorus dynamics is the accumulation and release of phosphorus from the bottom sediments. In a physical sense, solids may influence sediment-water exchange via burial. Additionally, the chemical composition of allochthonous particulate matter can have a decided effect on sediment feedback (Armstrong, 1979). Finally, the transport and fate of pollutants other than phosphorus are inextricably tied to solids. For example, many organic toxicants are extremely hydrophobic and when introduced into a lake tend to associate with particulate matter. The accurate modeling of these substances therefore requires that adequate information on the system's solids' budget be obtained.

In a more general sense, the foregoing analyses have been intended to caution against applying phosphorus loading models to systems where they are inappropriate. To date there have been numerous cases where empirical models developed from well-mixed lakes have been applied to systems as diverse as embayments, the coastal zone, and brackish estuaries. It is hoped that the present paper will prevent such misapplications in the future by showing how these systems are fundamentally different from completely mixed water bodies. Additionally, it is hoped that by demonstrating the importance of solids to modeling phosphorus dynamics, this paper represents a step toward improving these models in the future.

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THE APPLICATION OF THE LAKE EUTROPHICATION GAME SSWIMS TO THE MANAGEMENT OF LAKE GEORGE, NEW YORK

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ABSTRACT

During the past decade, limnologists have refined several concepts and techniques for managing freshwater lakes. In this paper, we will present a computer-based game based on such concepts as annual phosphorus and hydrologic budgets and empirical relationships among such indicator variables as winter total phosphorus, summer chlorophyll *a*, summer Secchi disk depth, extent of macrophyte growth, and hypolimnetic oxygen depletion. Although the game was originally designed as a vehicle for instructing personnel of government agencies in New York State in the subject of lake management, we shall discuss the use of the game in projecting the future quality of Lake George in Warren County, N.Y. under varied assumptions concerning changes in land use and wastewater treatment.

INTRODUCTION

The dynamics of lake ecosystems are generally too complex to understand through simple explanation. A number of North American researchers are using simulation modeling to more clearly understand biological, physical, and chemical relationships in lakes. This paper demonstrates the use of a simulation gaming model developed to help personnel of New York State agencies understand the consequences of various decisions which affect lake ecosystems. Lake George, in northern New York State, was selected as the study area to demonstrate the use of the model (Figure 1).

The simulation model consists of difference equations for lake winter total phosphorus, bottom anoxia, and extent of rooted vegetation. Regression equations developed for New York State lakes are used to calculate average summer Secchi disk depth and chlorophyll *a* from winter total phosphorus. Once the geomorphometric parameters of a specific lake are given to the program, the user may vary human population growth, land use patterns, and degree of treatment for sanitary wastes or stormwater. The SSWIMS model then predicts phosphorus, Secchi disk, chlorophyll *a*, bottom water oxygen, and the growth of rooted aquatic plants over any specified time period. Results may be displayed in either tables or plots. Variables in the game or the actual lake may be easily changed. The model program is written in FORTRAN, is inexpensive to use, and is designed for interactive use.

The conceptual model used for deriving the equations is shown in Figure 2. Its compartments represent either algebraic or difference equations. The

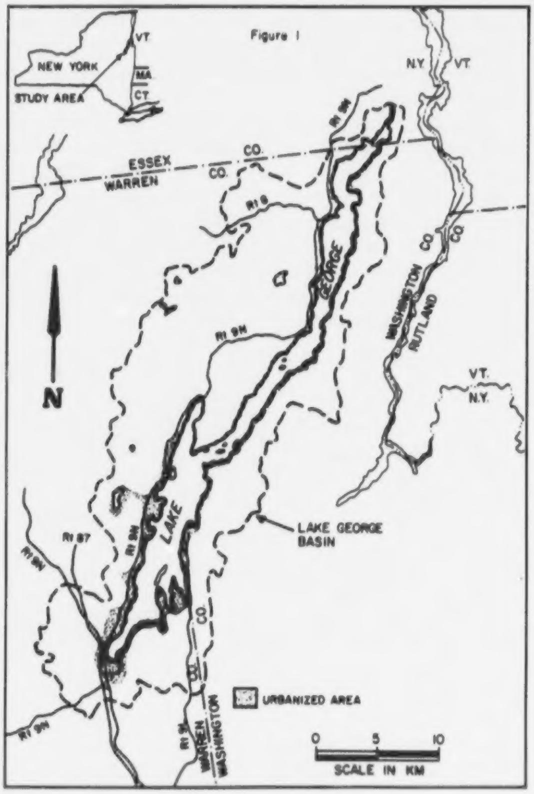


Figure 1. —

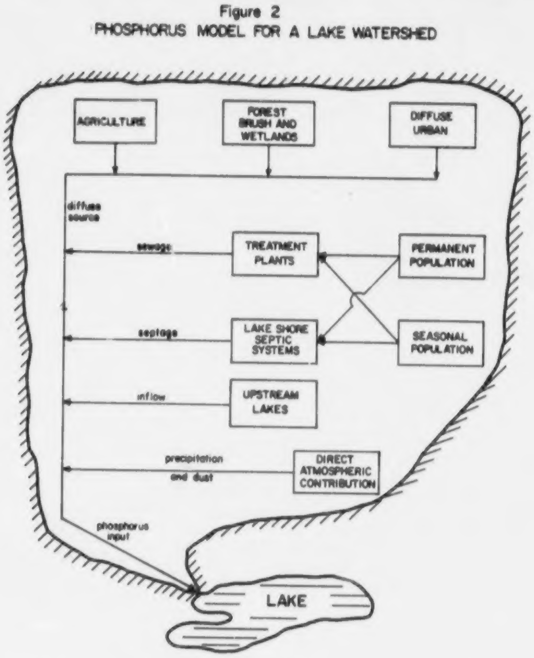


Figure 2
PHOSPHORUS MODEL FOR A LAKE WATERSHED
AGRICULTURE
FOREST BRUSH AND WETLANDS
DIFFUSE URBAN
diffuse source
sewage
TREATMENT PLANTS
PERMANENT POPULATION
septage
LAKE SHORE SEPTIC SYSTEMS
SEASONAL POPULATION
inflow
UPSTREAM LAKES
precipitation and dust
DIRECT ATMOSPHERIC CONTRIBUTION
phosphorus input
LAKE

Figure 2. — Phosphorus model for a lake watershed.

paths represent influence rather than transfer of an entity such as energy or matter. The direction of the arrow on each path indicates the active and passive compartments.

The major component of the SSWIMS model is an annual phosphorus budget (see Figure 3). Loadings from various sources in the watershed are added to the lake which is assumed to be well mixed. A certain fraction of the phosphorus is retained in the lake yearly, and under anoxic conditions, some is released from bottom sediments. The portion of the model related to fish production and quality of fishing is discussed only briefly in this paper.

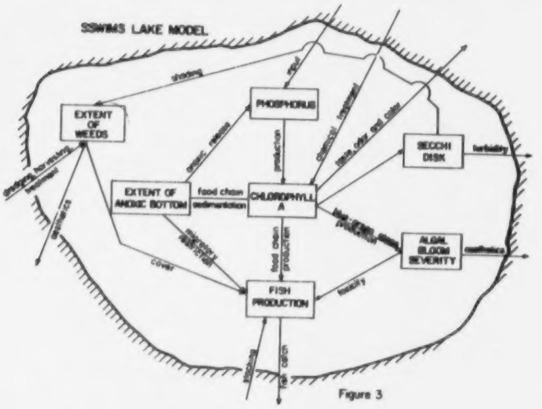

SSWIMS LAKE MODEL
shading
input
PHOSPHORUS
EXTENT OF WEEDS
organic release
production
chemical treatment
taste odor and color
SECCHI DISK
turbidity
dredging harvesting treatment
aesthetics
EXTENT OF ANOXIC BOTTOM
food chain
sedimentation
CHLOROPHYLL A
ALGAL BLOOM SEVERITY
aesthetics
cover
habitat restriction
food chain production
toxicity
FISH PRODUCTION
stocking
fish catch
Figure 3

Figure 3. — SSWIMS lake model.

THE MODEL

The model SSWIMS described here is a deterministic version lacking fishery equations. Our experience with driving the model with stochastic climatic inputs (amount of precipitation and runoff) has not improved the validity of the predictions, as continuous time series for variables such as winter total phosphorus or chlorophyll *a* do not exist for Lake George (Ferris, et al. in press).

A. *Phosphorus*

The phosphorus equation predicts the annual change in the amount of lake phosphorus (X_1). Its form is:

$$\frac{dX_1}{dt} = (RLOAD + RELS + HSUM + XNPSUM) - \left(\frac{1 - CUP{*}VOL}{TH} + FCHAIN\right){*}X_1 \qquad \text{eq. 1}$$

where:

CUP = Phosphorus concentration in inflow from upstream lake(s) (mg/m^3)
RLOAD = Direct atmospheric loading to the lake (mg/yr)

and

$$RLOAD = CPR{*}\ PR{*}\ AREAL \qquad \text{eq. 2}$$

where:

CPR = Phosphorus concentration in precipitation (mg/m^3)
PR = Annual precipitation (m^3/m^2)
AREAL = Lake surface area (m^2)
RELS = Anoxic release from bottom sediments (mg/yr.)

and

$$RELS = ALPHA{*}\ VOLAN{*}\ CHK/VOL \qquad \text{eq. 3}$$

where:

ALPHA = Anoxic release rate parameter (mg/yr.)
VOLAN = Summer volume of anoxic water (m^3) (see Section D)
VOL = Volume of lake (m^3)

and

$$CHK = 1 - BETA{*}\ X_1/VOL \qquad \text{eq. 4}$$

where:

BETA = Concentration dependent release parameter (m^3mg)
HSUM=Phosphorus contribution of septage and sewage (mg/yr.)

where:

$$HSUM = \sum_{i=1}^{n} GAMMA_i{*}(HP_i(T) + HS_i\,(T)/3))$$

and

GAMMA$_i$ = Unit load per capita for treatment level; (mg/cap-yr)
HP$_i$(T) = Permanent human population served by treatment level$_i$ at time T

$HS_i(T)$ = Seasonal human population (3 months) served by treatment level i at time T.

$HP_i(T)$ and $HS_i(T)$ are functions of time. For $HP_i(T)$, the function is :

$$HP_i(T) = HP_i(O) * (1 - PPI_i)^{T-TFIRST} \qquad \text{eq. 5}$$

where:

$HP_i(T)$ = Population at any time T.
$HP_i(O)$ = Initial population at time TFIRST.
PPI_i = Fractional annual population increase
TFIRST = Initial year
XNPSUM = Phosphorus contribution from diffuse sources (mg/yr)

where:

$$XNPSUM = AWS \sum_{j=1}^{m} XL_i * PLU_i/100 \qquad \text{eq. 6}$$

and

AWS = Watershed area (m^2)
XL_i = Unit load of phosphorus from ith land use (mg/m^2-yr)
PLU_i = Percent of watershed in the ith land use (unitless)

% Urban land use is increased using the same function as human population.

As urban use increases, forest and agriculture are decreased, but the ratio between % forest and % agriculture remains constant (an assumption).

FCHAIN = Exponential loss rate of phosphorus to food chain and bottom sediment (yr^{-1})
TH = Hydraulic retential time of lake (yr)

The annual amount of lake phosphorus is converted to spring total phosphorus concentration by the equation (Chapra and Tarapchak, 1976):

$$TP_{sp} = \left(\frac{X_1}{0.9}\right) * VOL \qquad \text{eq. 7}$$

B. *Chlorophyll a:*

Summer chlorophyll *a* (CHLOR) can be interpreted as a simple estimate of lake food chain production. The algebraic equation in SSWIMS is:

$$CHLOR = EXP\,(-0.51 + 0.86 * LN\,(TP_{sp})) \qquad \text{eq. 8}$$

C. *Secchi disk depth*:

Secchi disk depth (SECCHI) is a rough measure of water clarity. The algebraic equation is:

$$SECCHI = \left[\begin{matrix} 10.09 - 2.93 * LN\,(CHLOR) & \\ 0.10 & \\ & CHLOR \leqq 30\ mg/m^3 \\ & CHLOR > 30\ mg/m^3 \end{matrix}\right] \qquad \text{eq. 9}$$

Equations 8 and 9 are derived from data for New York State lakes (Oglesby and Schaffner, 1975, 1978; Bloomfield, 1978 a,b, 1980) concerning summer chlorophyll *a*, Secchi disk depth, and winter total phosphorus concentration. The data of Wood and Fuhs (1979) concerning Lake George tend to follow these relationships.

D. *Extent of Anoxic Conditions*:

A deficiency of oxygen in the bottom waters of a lake during summer thermal stratification generally indicates intense oxidation of organic materials in the bottom sediments and adjacent waters. A lack of oxygen in bottom waters during the summer is significant for two reasons.

First, reducing conditions tend to increase the solubility of phosphorus compounds, e.g., phosphorus in lake bottom sediments may dissolve and thus become available to stimulate algal and other plant growth. Second, anoxic conditions often lock valuable game fish into the cold bottom waters during the summer.

A simple difference equation based on the work of Welch and Perkins (1979) is used to simulate summer hypolimnetic oxygen depletion. The simulated variable (X_2) represents the areal hypolimnetic oxygen depletion rate. The equation is:

$$\frac{dX_2}{dt} = THETA * (ODR - X_2) \qquad \text{eq. 10}$$

where:

THETA = Decomposition rate constant (unitless)
ODR = Equilibrium depletion rate (mgO_2/m^2-day)

The equilibrium rate (ODR) is defined from Welch and Perkins (1979), in our notation:

$$ODR = 38.02 \left[\left(\frac{X_1}{AREAL}\right) * (1 + FC * TH) \right]^{0.37} \qquad \text{eq. 11}$$

where all constants and variables have been previously defined.

Hypolimnion dissolved oxygen at the end of summer thermal stratification is then defined as:

$$DOHPO = \begin{cases} DOSAT - \left(\dfrac{ANMAX * AHYPO * X_2}{VHYPO * 1000}\right) & DOHYPO < DOSAT \\ 0 & DOHYPO \geq DOSAT \end{cases} \qquad \text{eq. 12}$$

DOHYPO = Average hypolimnetic dissolved oxygen at the ($mg\ O_2/l$) end of summer thermal stratification
ANMAX = Maximum duration of thermal stratification (days)
VHYPO = Volume of hypolimnion (m^3)
AHYPO = Area of hypolimnion (m^2)
DOSAT = Oxygen saturation value for hypolimnion ($mg\ O_2/l$)

VHYPO and AHYPO are calculated from the cumulative volume and area functions and the following equation which was developed assuming a simple relationship between minimum depth of anoxia (ZANX) and hypolimnetic dissolved oxygen:

$$ZANX = ZST * \left(1 - \frac{DOHYPO}{DOSAT}\right) + ZBOT * \left(\frac{DOHYPO}{DOSAT}\right) \qquad \text{eq. 13}$$

where:

ZANX = Minimum depth of anoxia (m)
ZST = Depth of seasonal thermocline (m)
ZBOT = Depth of bottom (m)

and VOLAN, the anoxic volume (m^3) is then calculated from the cumulative volume relationship and ZANX.

E. *Macrophyte Growth*

The dynamics of macrophytes (aquatic weeds) in lakes have not been studied in enough detail to permit quantitative simulation. The term "weeds" in this paper will be limited to emergent and submergent vascular plants and macroalgae such as *Nitella*. The difference equation describing the area of the lake covered by a discernible weed growth (X_3) is relatively straightforward and has several assumptions implicit in its formulation. They are:

1. For a specific lake, weed beds can only increase to cover an ultimate area (AWMP, m^2). This potential area is defined by the morphometry of the lake and the fertility of the bottom sediments.
2. Light is a major limiting factor to aquatic weed growth.
3. Weed beds cannot extend into anoxic zones or into areas of poor growing conditions (extreme hydrostatic pressure, poor bottom conditions, high current activity, etc.) The equation for weed growth is:

$$\frac{dX_3}{dt} = \mathrm{WGROW} \cdot X \cdot (1 - X_3/\mathrm{AWMP}) \qquad \text{eq. 14}$$

where:

WGROW = Intrinsic rate of increase of weed beds (yrs^{-1})
AWMP = Area of potential weed penetration (m^2)

and:

$$\mathrm{AWMP} = \mathrm{AREAL} \cdot (1 - F(\mathrm{ZWMP})) \cdot \mathrm{WPER} \qquad \text{eq. 15}$$

where:

$$\mathrm{ZWMP} = \left(\begin{array}{c} 0.83 + 1.22 \cdot \mathrm{SECCHI} \\ \mathrm{ZANX} \end{array} \right) \begin{array}{l} \mathrm{ZWMP} < \mathrm{ZANX} \\ \mathrm{ZWMP} \geq \mathrm{ZANX} \end{array}$$

(Dunst, in press) eq. 16

ZWMP = Depth of potential weed penetration (m^2)
WSECC = Light dependent coefficient (unitless)
WPER = Percent of total lake area where weeds will grow potentially (unitless)
F(ZWMP) = Depth, vs bottom area relationship for a specific lake

APPLICATION OF SSWIMS TO LAKE GEORGE

Lake George is located in the eastern Adirondack Mountains of New York State and the southeastern portion of the Adirondack State Park (Figure 1). It is one of the most heavily used recreational waters in the eastern United States. Most homes around the lake use its water directly, often without disinfection.

Recognizing that the lake is a unique resource in the northeastern United States, there has been a strong, long-term State and local commitment to protect and enhance water quality in Lake George. However, efforts to protect water quality in Lake George have not been entirely successful. Since the early 1920's when Secchi disk measurements were first published, water transparency has decreased from 10 to slightly over 6 meters (Needham, et al. 1922).

Recent efforts to stem and reverse the trend toward eutrophy and increasing bacterial pollution in Lake George have culminated in a plan to collect and divert sewage out of the south basin of the lake. This proposal, which enjoys strong support among many lake residents and government officials, is not without its critics who argue that sewering the lake will facilitate its development. Critics also contend the increased storm runoff from an expanding urban area will more than offset the benefits of sewering the south lake basin.

The interaction of these factors — population growth, sewering vs. non-sewering, and controlling phosphorus in urban storm drainage — were compared using the SSWIMS model. Fourteen scenarios with a time frame to the year 2030, which corresponds to the project design capacity of the proposed Warren County sewer system, were simulated.

Examining three different population projections revealed that the annual rate of population increase in the southern basin of Lake George is about 1.4 percent. An annual rate of increase of 2.0 percent served as the upper limit of population growth on the assumption that sewering the lake basin would speed up the rate of development.

It was also assumed that urban expansion in the lake basin would increase in proportion to population growth, or that doubling the population would double the urbanized area. Management policies for controlling 25, 50, and 90 percent of the phosphorus in urban storm runoff were compared with each other and with a policy of non-control.

The value of each parameter and constant used in the simulation is shown in Tables 1 and 2. Their sources are:

1. Physical Constants (TH, VOL, AREAL, AWS, ZBOT). These values were obtained from Wood and Fuhs (1979) and agree closely with other published information on Lake George.
2. Land use parameters (XLU, PLU) were developed from Fuhs (1972) and Hetling (1974). Since agriculture and urban area is presently quite limited in the Lake George watershed, unpublished information provided by Nicholas L. Clesceri (pers. comm.) on stormwater quality was used to confirm that the phosphorus loading estimates made by Hetling (1974) were reasonable.
3. Per capita phosphorus contribution parameters (GAMMA) were assumed to be zero for the sewered population. The sewered areas are served by two small municipal plants at Lake George Village and Bolton Landing. Each plant discharges to natural sand beds and extensive field work has indicated that these plants probably contribute less than 5 percent of the total annual input of phosphorus to Lake George (Aulenbach, et al. 1976). The value of GAMMA for the population served by septic tanks was determined from the estimates of septic tank phosphorus contributions made by Gibble (1974), Hetling (1974) and Ferris, et al. (in press). None of these estimates has been checked by field measurements.

Table 1. — Parameters and constants representing conditions used in simulations of South and North Lake George.

SYMBOL	PARAMETER OR CONSTANT		SOUTH	NORTH
TH	Hydraulic retention time of lake	yr	6.90	4.49
VOL	Volume of lake	m^3	1.02×10^9	1.08×10^9
AREAL	Surface area of lake	m^2	5.8×10^7	5.6×10^7
AWS	Watershed area	m^2	3.1×10^8	1.8×10^8
ZBOT	Maximum depth	m	58.0	53.0
FCHAIN	Phosphorus retention rate	yr^{-1}	0.28	0.25
THETA	Decomposition rate parameter	unitless	1.0	1.0
DOSAT	Hypolimnetic saturation dissolved oxygen	$mg\text{-}liter^{-1}$	13.6	13.6
ALPHA	Anoxic release rate parameter	$mgP\text{-}yr^{-1}$	1.0×10^9	1.0×10^9
BETA	Concentration dependent release parameter	$m^3 mgP^{-1}$	0.005	0.005
ZST	Depth of seasonal thermocline	m	12.0	12.0
WGROW	Aquatic vegetation growth rate	yr^{-1}	0.5	0.5
WPER	Potential percent of lake where vegetation will grow	unitless	50.0	50.0
ANMAX	Maximum duration of thermal stratification	days	150.0	150.0
PPI	Growth rate parameters for human population groups and developed area	yr^{-1}	0.014	0.014
CPR	Phosphorus concentration in precipitation	$mgP\text{-}m^{-3}$	10.0	10.0
PR	Annual precipitation	m	1.0	1.0

Table 2. — Parameters and constants related to human population and land use.

LAND USE	PLU (percent) South	PLU (percent) North	XLU $mgP\text{-}m^{-2}yr^{-1}$)
Forest	82	95	4.0
Cropland	5	0	30.0
Developed	3	2	100.0
(Non-contributing)	(10)	(3)	(0.0)

TREATMENT TYPE	Population Served (1975)				GAMMA
	South Basin		North Basin		$(mgP\text{-}cap^{-1}\text{-}yr^{-1})$
	HP	HS	HP	HS	
Sewered, no discharge	2,200	21,500	0	0	0.0
Septic tanks	2,900	26,400	1,100	3,300	6.0×10^4

PLU — Percent of watershed in ith land use
XLU — Unit load of phosphorus from ith land use
HP — Permanent human population served by treatment level i
HS — Seasonal human population served by treatment level i

4. Phosphorus retention (FCHAIN) was determined by dividing the difference between annual phosphorus inputs and losses (outflow) by the annual average phosphorus concentration of the lake water. Various estimates of phosphorus retention for Lake George made by Aulenbach (1973), Ferris, et al. (in press) and Wood and Fuhs (1979) yield values for FCHAIN ranging from 0.2 yr to over 1.5 yr .

5. Anoxic zone and aquatic vegetation parameters (THETA, ALPHA, BETA, ZST, WGROW, WPER, ANMAX) were estimated using SSWIMS data from a variety of New York State lakes and from Ferris, et al. (in press) data concerning vegetation, thermal stratification, and dissolved oxygen. Lake George's hypolimnion is at present well oxygenated and aquatic vegetation often is 9 to 10 meters deep.

6. Precipitation parameters (PR, CPR). The annual average precipitation was estimated from U.S. Department of Commerce data (1979) for the Glens Falls, N.Y. station. The phosphorus content of precipitation was derived from Wood and Fuhs (1979) and from 10 years of data from the New York State precipitation chemistry network (U.S. Dep. Inter. 1979).

7. Human population estimates (HP, HS) and projections were derived from Lawler, Matusky, and Skelly, Inc. (1975), New York Department of Environmental Conservation (1976), and Hazen and Sawyer (1977). Annual human population growth is estimated at an average of 1.4 percent.

RESULTS

Figure 4 shows that of the 14 water quality management alternatives compared with the model, only six maintain existing water quality or increase water transparency in Lake George. Of these, four are not feasible because they would require more than 25 percent removal of phosphorus from urban storm runoff. The lack of suitable terrain for constructing control devices makes these alternatives technically difficult to accomplish (Figure 4). Therefore, only 2, 6, and 13 appear technically feasible.

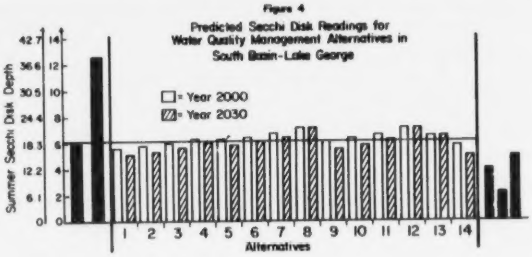


Figure 4. — Predicted Secchi disk readings for water quality management alternatives in south basin-Lake George.

Under alternative 13 (Table 3), population growth would have to be limited and urbanization held to present levels. Also, the proposed sewer system would not be constructed and phosphorus in urban runoff

Table 3. — Limit population growth and halt urban development, no sewerage system (south basin).

Year	Win Tot P (μg/l)	Chlor *a* (μg/l)	Secchi	Z Anoxic meters	Bloom Sev.	% Area Weeds	Population	% Basin Urban
1975	9.	4.	6.1	57.4	None	13.6	53,000	3.
1980	8.	4.	6.4	57.0	None	14.4	53,000	3.
1985	8.	3.	6.4	57.0	None	14.7	53,000	3.
1990	8.	3.	6.5	57.0	None	14.7	53,000	3.
1995	8.	3.	6.5	57.0	None	14.7	53,000	3.
2000	8.	3.	6.5	57.0	None	14.7	53,000	3.
2005	8.	3.	6.5	57.0	None	14.7	53,000	3.
2010	8.	3.	6.5	57.0	None	14.7	53,000	3.
2015	8.	3.	6.5	57.0	None	14.7	53,000	3.
2020	8.	3.	6.5	57.0	None	14.7	53,000	3.
2025	8.	3.	6.5	57.0	None	14.7	53,000	3.
2030	8.	3.	6.5	57.0	None	14.7	53,000	3.

Table 4. — Construct sewer, reduce P in urban storm runoff by 25 percent. 1.4 percent annual growth rate (south basin).

Year	Win Tot P (μg/l)	Chlor *a* (μg/l)	Secchi	Z Anoxic meters	Bloom Sev.	% Area Weeds	Population	% Basin Urban
1975	9.	4.	6.1	57.4	None	13.6	53,000	3.
1980	7.	3.	6.6	57.0	None	14.7	56,815	3.
1985	7.	3.	6.6	57.0	None	14.9	60,905	3.
1990	7.	3.	6.5	57.0	None	14.9	65,290	4.
1995	8.	3.	6.5	57.0	None	14.8	69,990	4.
2000	8.	3.	6.4	57.0	None	14.7	75,028	4.
2005	8.	4.	6.4	57.0	None	14.6	80,429	5.
2010	8.	4.	6.3	57.0	None	14.5	86,219	5.
2015	8.	4.	6.3	57.0	None	14.4	92,426	5.
2020	8.	4.	6.2	56.9	None	14.3	99,080	6.
2025	9.	4.	6.2	56.9	None	14.1	106,212	6.
2030	9.	4.	6.1	56.9	None	14.0	113,858	6.

would not be controlled. Such indicators as predicted nutrient enrichment, chlorophyll *a*, Secchi disk depth, depth to anoxic conditions, bloom severity, and percent area of the lake supporting weed growth, as revealed in Table 3, show that as long as this policy remains in effect the lake will enter into a steady state equilibrium in which there would be no further impairment to water quality as measured by these indicators. Although water quality in Lake George could be maintained at present levels for the indefinite future under this alternative, there would be no net improvement in water quality.

As Figure 4 and Table 4 reveal, water quality can be maintained to the year 2030 by sewering the south lake basin and reducing phosphorus in urban storm runoff by 25 percent, provided that population growth and urbanization do not increase above the current projected annual rate of 1.4 percent (alternative 6).

An examination of Figure 4 reveals, however, that if the annual growth in the south basin of Lake George is allowed to increase to 2.0 percent, constructing the sewer and reducing P in urban storm runoff by 25 percent as in alternative 9, is not sufficient to offset the impact of accelerated growth and development.

DISCUSSION

An examination of the simulated scenarios reveals relatively few options either for enhancing water quality in the south basin of Lake George or maintaining it at existing trophic levels. As the predicted Secchi disk readings for the south basin in Figure 4 show, only two water quality management alternatives, 6 and 13, achieve this goal while appearing to be technically feasible. The remaining alternatives either fall short of maintaining present levels of water quality in the south basin, or would be technically difficult to accomplish.

Although technically sound, alternative 13 is probably neither economically or politically feasible. This alternative would require an immediate cessation of growth and development in the south basin. Presumably, strategies to limit growth would require combining strict land use controls with regional growth-inhibiting economic policies.

The findings do indicate that water quality can be maintained at present levels to the year 2030 through sewering the south lake basin and reducing P in urban storm runoff by 25 percent, provided that population growth and urbanization do not increase above the current annual rate of 1.4 percent (alternative 4). With a modest investment in control structures, backed by land use planning, management, and controls, it appears technically feasible to reduce P in urban storm runoff by 25 percent.

There is, however, considerable uncertainty as to the influence the sewer system will exert on rates of growth and development in the south basin. Some have suggested that the sewer system will accelerate the rate of population growth and urban development. If, for any reason, the annual rate of growth and development exceeds 1.4 percent, the policy of sewering the south lake basin and of reducing P in urban storm runoff by 25 percent will fall short of meeting the objective of maintaining the present level of water quality for the life of the sewer project, i.e., to the year 2030 (see Figure 4).

A further implication is that, if it is concluded that reducing P by more than 25 percent is not feasible and if the sewer project accelerates development, then controls which curb growth and development would have to be instituted before 2030.

Faced with considerable uncertainty about future rates of growth (if the goal is to maintain or enhance water quality), prudent decisionmaking would dictate that a water quality management strategy should be based on a 2 percent annual rate of growth. Growth would have to be limited to achieve the goal unless a much greater control of P could be insured.

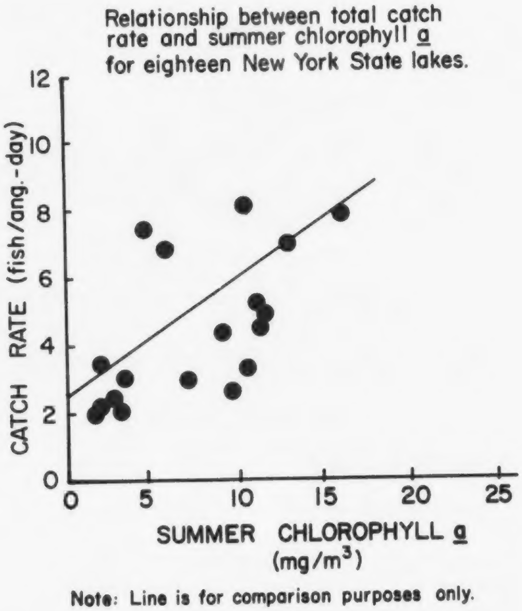
Other than simulating in-basin tertiary treatment with 90 percent P removal (alternative 14), no simulations were made for other in-basin wastewater treatment alternatives in this paper. However, what is clear from the SSWIMS simulations is that no matter what the alternative for in-basin treatment may be, it must provide almost 100 percent P removal, combined with 25 percent P reduction in urban runoff if water quality is to be maintained at present trophic levels. Furthermore, in terms of a comprehensive approach to water quality planning, in-basin strategies to effect wastewater treatment may conflict with strategies for reducing P in urban storm runoff. For example, diverting wastewater outside the basin, as is currently proposed, would make the sand filter beds at the Bolton and Lake George Village sewage treatment plants available for treatment of storm runoff. Presumably, most approaches to in-basin wastewater treatment would use the sand filter beds, thereby preventing their potential use for treating urban runoff.

Figure 4 also contains some additional water clarity information for comparison. On the far left are the present average summer Secchi disk depths for the north and south basins of Lake George as predicted by SSWIMS. These values agree with information presented by Clesceri, et al. (in press) and Wood and Fuhs (1979). On the right of the diagram are the average summer Secchi disk depths for three New York State lakes with a morphometry similar to Lake George: Canandaigua Lake in western New York, Cayuga Lake, a Finger Lake impacted by point sources or municipal waste, and Otsego Lake in east-central New York. All presently exhibit water clarity inferior to almost all of the scenarios predicted by SSWIMS for South Lake George. This is presented only to show how unique the present condition of Lake George is when compared to other major New York State lakes.

THE QUESTION OF FISHERIES MANAGEMENT

The original SSWIMS model included State variables for management of lake fish community and associated sport fisheries. Although on a global basis, Ryder's Morphoedaphic Index (MEI, Ryder, et al. 1974) and the more recent work of Oglesby (1977) have been used to predict fish production from morphometry and indicators of lake trophic status, it was clear to us at that time that the information available for New York State lakes was not sufficient to test the validity of either technique. This is still true because of the extreme effort required to accurately estimate fish numbers in a large lake. The work of David Green on the fishery of Canadarago Lake (Harr, et al. 1980) is one of the few efforts documented in which changes in the fish community were quantified concurrently with limnological studies and improved wastewater treatment measures. Although phosphorus inputs to the lake were significantly reduced in 1973, only minor fluctuations in water clarity, chemistry, and biota occurred. However, major changes in the numbers of various fish species were observed, probably related to climatic conditions. It should be noted that reducing phosphorus input to the lake did not affect the MEI parameters.

Thus, the inclusion of fishery variables in eutrophication models remains limited by the future collection of adequate fisheries data. However, recent comparisons of data in New York State Angler Surveys (Brown, 1973; NYS Dep. Environ. Conserv. 1978) with trophic status of various lakes reveal that the total catch per unit effort (Figure 5) and the relative proportion of gamefish to forage species taken (Figure 6) are somewhat related to such variables as summer chlorophyll *a*. Ideally, the model should provide outputs on the resulting fisheries and their associated socioeconomic values. The challenge of meshing fishery and limnological concepts lies in relating catch per unit effort of various fish species to productivity indices. In turn, these catches per unit of effort could be translated into socioeconomic values which would help decisionmakers in determining the best alternative. The information from this segment of the model would also be useful in assessing the effectiveness of the various management techniques, such as seasons, creel limits, size limits, habitat improvement, and



Note: Line is for comparison purposes only.

Figure 5. — Relationship between total catch rate and summer chlorophyll *a* for 18 New York State lakes.

stocking (see Figure 7). Perhaps the best we can hope for at this time is to be able to project whether the fishing (fish/angler-day) for a given species will be good, fair, or non-existent, and subsequently weigh the socioeconomic impacts. Nevertheless, the challenge to do better should not be ignored.

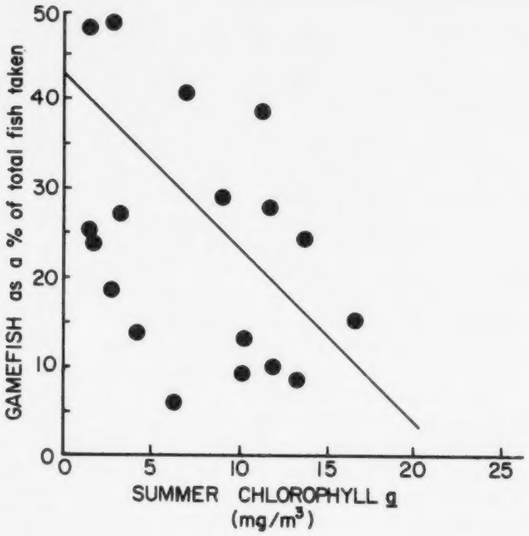


Figure 6. — Relationship between gamefish taken as a percentage of total angler catch and summer chlorophyll *a* for 18 New York State lakes.

Notes: 1) Line is for comparison purposes only.
2) Gamefish are black bass, walleye, all esocids and salmonids.

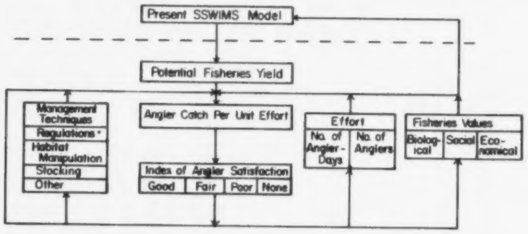


Figure 7. — Conceptual fisheries elements for the SSWIMS lake management model.

*Regulations could involve seasons, creel limits, sizes, gear, locations or combinations of them.

CONCLUSIONS

The SSWIMS model is a useful predictive tool providing decisionmakers with insights to more clearly understand the relationships between water quality management variables and the consequences of policy decisions on water quality.

While it cannot be presumed that the data used in the SSWIMS model are precise, indications are, that to maintain or enhance water quality in the south basin of Lake George, a strategy of sewering the south basin and instituting land management controls designed to reduce P in urban storm runoff must be employed, as in alternative 6. It would appear, on the basis of the data used, that this strategy may have to be reinforced with a policy of limiting growth and development. One way to accomplish this would be to orchestrate a system of land use controls with an interceptor pipe sized to a specific capacity consistent with the desired population ceiling.

The SSWIMS model is not a substitute for decisionmaking; rather, the simulated outcomes should be used to guide the decisionmaking process. Armed with a knowledge of the potential consequences of their decisions on Lake George, policymakers have choices that reflect appropriate levels of growth and development, interceptor pipe size, and control of P in urban storm runoff in relation to water quality management objectives.

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VARIABILITY OF TROPHIC STATE INDICATORS IN RESERVOIRS

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ABSTRACT

As part of the Environmental Water Quality Operation Studies being conducted by the Army Corps of Engineers, a data base has been compiled that describes the morphometry, hydrology, and water quality of over 300 reservoirs throughout the United States. The data base will be used to test and evaluate existing empirical models for assessing eutrophication problems and to develop new methods, where appropriate. This work has been motivated by concerns over the application of existing models to reservoirs, despite the fact that most have been developed using data bases consisting entirely of northern, natural lakes. Existing methods may not be adequate for reservoirs because of differences in morphometry, hydraulics, sedimentation, and region, that may influence responses to nutrient loading. To provide preliminary insights into the effects of using different data-reduction procedures and into the adequacy of the data for model testing purposes, EPA National Eutrophication Survey data from 76 phosphorus-limited Corps impoundments are analyzed and used in testing Carlson's (1977) Trophic State Indices. Seasonal effects and variance/covariance components are identified at different averaging levels. Results indicate that chlorophyll *a* levels in Corps reservoirs are generally less sensitive to phosphorus or transparency than in the natural lakes used by Carlson in developing the index system. The use of error analysis for assessing the adequacy of the data set for model testing purposes is demonstrated.

INTRODUCTION

The development of phosphorus loading/trophic state response models over the past decade has greatly increased the feasibility of lake water quality planning. Most of these models have been based upon empirical studies of data from natural lakes in glaciated regions. Their applicability to manmade impoundments is in question because of lake/reservoir differences in age, morphometry, hydrodynamics, sedimentation, and region (Thornton, et al. 1980). To provide a basis for testing available models, data describing the morphometry, hydrology, water quality, and sedimentation rates of over 300 active U.S. Army Engineer reservoirs have been compiled (Walker, 1980a). During the next year, this data base will be used in a systematic assessment of phosphorus loading models and relationships among trophic state indicators in reservoirs.

The data base currently contains over two million water quality observations. Testing empirical eutrophication models in reservoirs requires averaging water quality measurements over spatial and temporal scales. If within-pool water quality variations are not random with respect to date, station, or depth, then summary statistics for a given reservoir will depend to some extent upon the particular data reduction method employed. The choice of reduction method may, in turn, influence conclusions regarding the adequacy of existing models as well as the parameter estimates of any new models which may be developed.

There is no standard data reduction procedure which can be used prior to model development, testing, or application. Methods have included, for example, (1) taking the median or mean of all within-pool observations (U.S. EPA, 1975); (2) sequential averaging over depths, stations, and dates (Lambou, et al. 1976); (3) sequential averaging within specific depth ranges (Carlson, 1977); and (4) various weighted averaging schemes which reflect morphometric characteristics (Boyce, 1973). As compared with natural lakes, many reservoirs pose special data reduction problems because of extreme spatial and/or temporal variations in conditions.

This paper describes investigations of the variability of trophic state indicators among and within a group of Corps reservoirs. The analysis covers seasonal relationships, variance/covariance components, regression analyses, and error analyses. This work has been undertaken to assess the implications of using different data reduction procedures and to assess the adequacy of the data for model testing purposes.

DATA BASE

National Eutrophication Survey (U.S. EPA, 1973) data have been used as a basis for this analysis. The relatively uniform sampling program designs used by the survey provide data that are suitable for statistical treatment. One drawback, however, is that under this program reservoirs were typically sampled only three times during one growing season. In future work, we plan to examine data from other agencies, which, in many cases, are more intensive and/or cover longer periods. The Survey data have been screened to eliminate data from 19 reservoirs which were predominately nitrogen-limited (based upon bioassays) and to eliminate all stations with fewer than three sampling dates for total phosphorus, chlorophyll *a*, and transparency. The resulting file contains 963 observations from 306 stations in 76 reservoirs.

Surface total phosphorus, Secchi depth, and chlorophyll *a* values have been expressed in terms of Carlson's Trophic State Indices (Carlson, 1977):

$$I_P = 4.2 + 33.2 \log_{10} P \qquad \text{eq. 1}$$
$$I_T = 60 - 33.2 \log_{10} Z_s \qquad \text{eq. 2}$$
$$I_B = 30.6 + 22.6 \log_{10} B \qquad \text{eq. 3}$$

where,

P = total phosphorus concentration (mg/m^3)
Z_s = Secchi depth (m)
B = chlorophyll *a* concentration (mg/m^3)
T = transparency

The indices are calibrated so that the three versions are equivalent, on the average, when applied to midsummer, epilimnetic data from northern, natural lakes. Expression of measurements on these scales tends to reduce the skewness in the distributions of the variables and provides benchmarks for assessing reservoir trophic state relationships in comparison to those typical of natural lakes.

The latitudes of 309 natural lakes sampled by the Survey are compared with the latitudes of 106 Corps reservoirs sampled by the Survey in Figure 1. The distribution of natural lakes is bimodal, with a northern peak (glacial lakes) and a southern peak (subtropical lakes in Florida). Most of the Corps reservoirs may be influenced by regional factors as well as the effects of impoundment type.

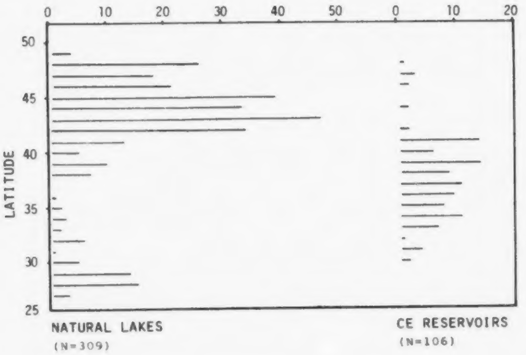

Figure 1. — Latitudes of natural lakes and Corps reservoirs sampled by the EPA National Eutrophication Survey.

SEASONAL RELATIONSHIPS

Average seasonal variations in the index components are depicted in Figure 2. Station means have been computed and their effects removed from the data prior to calculating the mean and standard error for each month (March to November) and index component. Analyses of variance indicate that fixed monthly effects are significant ($p < .0001$) but explain only 11 percent of the total within-station variance of each index. The seasonal variations depicted in Figure 2 are characteristic of this collection of reservoirs but not necessarily of each individual reservoir.

Average seasonal effects on phosphorus and transparency are similar: Both tend to be lowest during March and midsummer and highest during April and November, possibly reflecting seasonal flow and turbidity variations and the influences of turnover periods. Monthly effects on chlorophyll *a* suggest a spring maximum (April-May), followed by a June depression, a midsummer maximum, and lower values in November. Temperature and light effects may be responsible for the relatively low chlorophyll *a* levels during March and November. The June depression may be caused by seasonal succession of algal species. A more detailed examination of the data indicates that lower June chlorophyll *a* levels are characteristic of about half of the stations sampled in June, while the rest have June levels more typical of May or July values. In testing seasonal aspects of TSI behavior, Carlson (1977) also noted a June depression in chlorophyll *a* index relative to the phosphorus index in three natural lakes.

Differences among various versions of the index provide a measure of "lake-like" behavior, since the index system is calibrated so that I_P, I_T, and I_B, values are equivalent, on the average, when applied to midsummer epilimnetic data from northern, natural lakes. Figure 2 indicates that the range of index means is generally lowest during midsummer and highest during March, June, and November (approaching 15). Minor recalibration of the phosphorus and/or transparency index would bring I and I into agreement for all seasons, since the monthly effect curves in Figure 2 are roughly parallel. Since seasonal chlorophyll *a* behavior is fundamentally different, however, recalibration alone would not eliminate biases (i.e., significant differences between I_B and I_P or I_T) for all seasons.

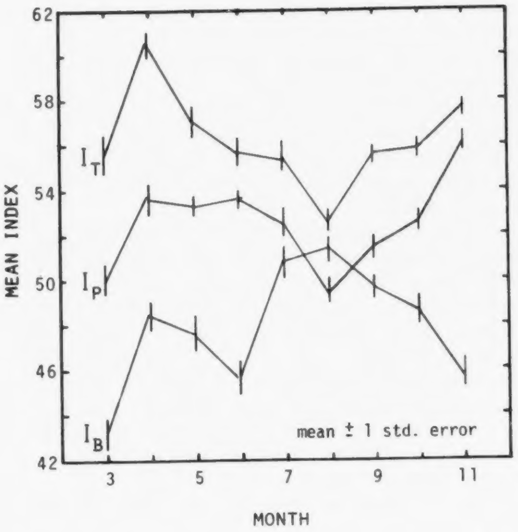

Figure 2. — Monthly variations in trophic state indices.

VARIANCE COMPONENTS

Trophic index observations can be classified in a hierarchy defined by region, reservoir, station, and sampling date. Variations at each level could account for some portion of the total variance of each index. A nested analysis of variance procedure (Statist. Anal. Inst. 1979) has been applied to derive pooled estimates of variance and covariance components according to the following model:

$$\mathrm{Var}(I) = \sigma^2_d + \sigma^2_{r(d)} + \sigma^2_{s(d,r)} + \sigma^2_e \qquad \text{eq. 4}$$

where,

σ^2_d = variance among regions, defined by Corps districts

$\sigma^2_{r(d)}$ = variance among reservoirs, within districts

$\sigma^2_{s(d,r)}$ = variance among stations, within reservoirs and districts

σ^2_e = variance within stations

This model has been used to describe variations in the data. It is of limited use for significance testing, which would require randomness and serial independence in the within-station variations, that can be attributed to variations in time, sampling error, and measurement error. As demonstrated in the previous section, some of the within-station variations can be attributed to seasonal factors and are therefore nonrandom. Given three observations per station spaced at roughly bimonthly intervals, serial dependence in the observations is not likely to be strong, since conditions are known to vary in many reservoirs at a much higher frequency, as influenced, for example, by storm events and algal bloom occurrences. Among-station, within-reservoir variations also show some serial dependence, since spatial trends in the indices are often apparent when station means are displayed in a downstream order (Walker, 1980b).

The relative magnitude of the last term is of special significance to modelling efforts. With relatively large within-station variance, it would be difficult to obtain much accuracy in station summary statistics (e.g., station mean) with limited data. This would reduce the explainable variance of any model or index system calibrated to the reduced data set, make it more difficult to distinguish among alternative model formulations, and increase the error associated with model parameter estimates.

Variance components estimated for each index are displayed on the left side of Figure 3. Variations in the phosphorus and transparency indices are similar at all levels. Variance components of the chlorophyll *a* index at the district, reservoir, and station levels are considerably lower than would be predicted based upon corresponding phosphorus and transparency variance components. The within-station components account for a major portion (~60 percent) of the total chlorophyll *a* variability. Thus, on the average for this data set, temporal variations in the chlorophyll *a* index at a given station appear to be stronger than variations among stations, reservoirs, and/or districts. The within-station variance components correspond to standard deviations of 6.5, 6.5 and 7.9 for I_B, I_T, and I_P respectively.

The covariance components on the right side of Figure 3 provide insights into relationships among the indices at different averaging levels. Spatial covariances are positive in all cases. Thus, the various versions of the index correlate positively among districts, reservoirs within districts, and stations within reservoirs. Appreciable temporal covariance is observed only for phosphorus and transparency. This covariance might be attributed, for example, to turbidity variations following seasonal or short-term (storm event) flow variations. Despite its positive covariance spatially, the chlorophyll *a* index does covary temporally with the other indices.

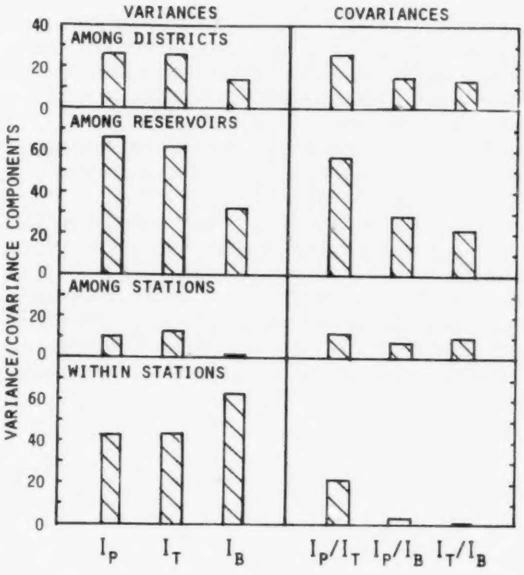

Figure 3. — Variance/covariance components of trophic state indices.

REGRESSION ANALYSES

The covariance components indicate that the indices can be averaged by station with some loss of information about the phosphorus/transparency relationship, but without losing chlorophyll *a* predictability. Figure 4 depicts relationships among station-mean values of the indices. Results of standard and geometric-mean regression analyses relating the indices are given in Table 1. Geometric mean regressions summarize functional relationships and are appropriate to use when both the independent and dependent variables are subject to natural variability and measurement error (Ricker, 1973). Standard regressions are appropriate for predictive purposes.

The phosphorus and transparency indices explain 37 and 29 percent of the variance in the chlorophyll *a* index, respectively. Recalibration of the index system to these reservoirs requires significant reductions in the corresponding slopes. In contrast, the phosphorus index explains 78 percent of the transparency index

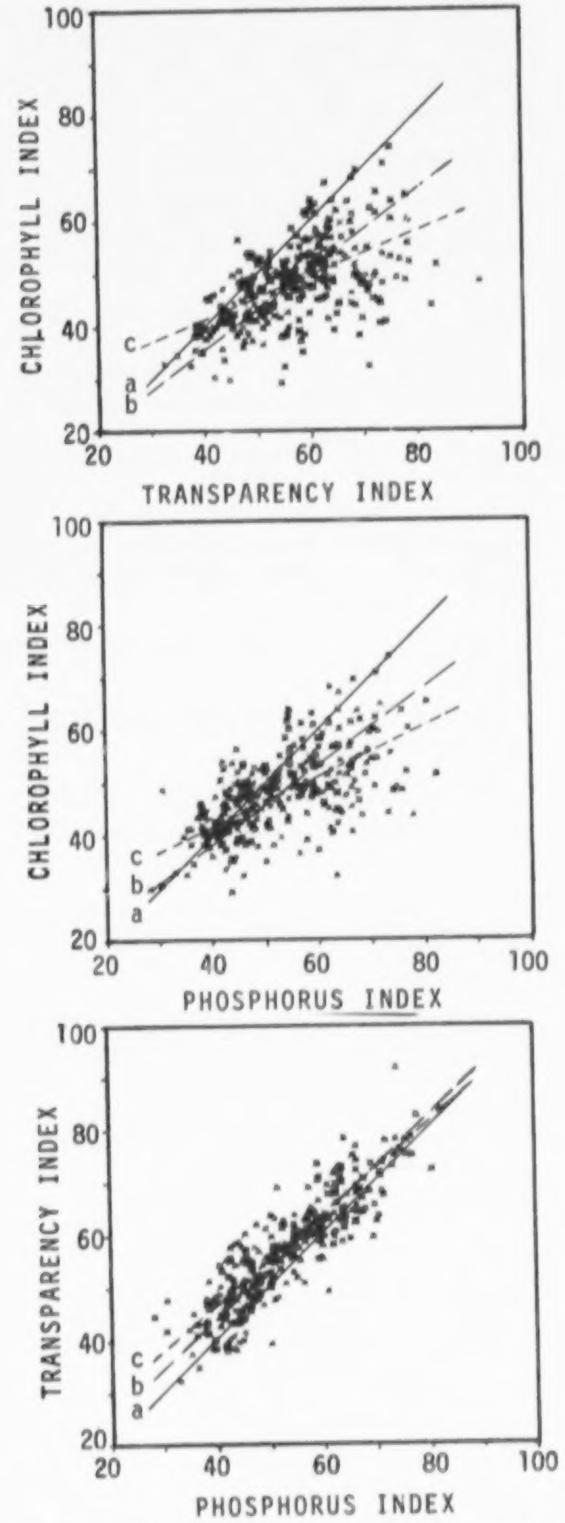

Figure 4. — Relationships among station-mean index values (a = line of equality, b = geometric-mean regression, c = standard regression).

variance and requires an adjustment in the intercept only. Thus, compared with chlorophyll *a*, the phosphorus/transparency relationship appears to be more typical of the natural lakes used by Carlson in deriving his index system.

The effects of using alternative data reduction procedures on the regression analyses have been also studied. Using only summer mean values reduces the regression slopes and R^2 values and increases mean squared residual errors by 58, 46, and 94 percent for the I_B/I_P, I_B/I_T, and $I_{T/P}$ regressions, respectively. These increases in error result partially from loss of within-station replication when spring and fall values are eliminated. In future work, data from other monitoring programs with more intensive summer sampling will be investigated. Use of reservoir means has little influence on the results, but increases the standard errors of parameter estimates.

ERROR ANALYSES

Residual errors from the regressions can be attributed to three types of error: parameter, data, and model. The first reflects uncertainty in the model coefficients; the second, errors in the predicted and/or predictor variables; and the third, influences of factors which are not considered in the model structure. The results of the variance component and regression analyses can be used to derive approximate estimates of the data errors according to the following equation:

$$\mathrm{Var}(R)_D \quad \frac{\mathrm{Var}(I_y)_E + b^2\mathrm{Var}(I_x)_E - 2b\,\mathrm{Cov}(I_y, I_x)_E}{N} \qquad \text{eq. 5}$$

where,

$\mathrm{Var}(R)_D$ = data-error component of mean-squared residual
$\mathrm{Var}(I_y)_E$ = within-station variance of predicted index
$\mathrm{Var}(I_x)_E$ = within-station variance of predictor index
b = slope of regression equation
$\mathrm{Cov}(I_y, I_x)_E$ = within-station covariance of predicted and predictor indices
N = number of observations per station (averaging 3.1)

This formula is approximate because it assumes serial independence in the within-station variations, which would tend to be more important at sampling intervals less than the 2-month intervals characteristic of this data set.

The results of applying this equation to the regression models in Table 1 are given in Table 2. They indicate that roughly half (50 to 59 percent) of the residual errors from the regressions can be attributed to data errors. These components could be reduced with a more intensive sampling program (i.e, more replications per station). The influences of parameter uncertainty on the total residual error are expected to be relatively insignificant, since the parameter error component is inversely proportional to the number of stations used in the regression analyses and the parameters are relatively well-determined (Walker, 1977). Thus, most of the remaining error can be attributed to the effects of factors which are not considered in the index system.

Since data errors do not explain all of the residual variance, it may be possible to improve the index system by modifying it to take other important factors into account. One modification is suggested by these results and by the turbid nature of many reservoirs. Chlorophyll *a*/phosphorus and chlorophyll *a*/transparency relationships may not be constant across reservoirs because of variations in non-algal particulate materials (turbidity), which would influence measurements of total phosphorus and transparency but not of chlorophyll *a*. The relative stability of the phosphorus/tranparency relationship across lakes and reservoirs may be attributed to the fact that both types of measurements are sensitive to algal and non-algal particulate materials. Other factors which might contribute to model error include kinetic effects in reservoirs with short hydraulic residence times. It might also be possible to modify the system to account for nitrogen limitation, by including N-limited as well as P-limited reservoirs in the data set. Expansion of the index system to include hypolimnetic oxygen deficits is another possibility (Walker, 1979). These approaches will be investigated in future studies of Carlson's index system and other schemes using more extensive and intensive data sets derived from the Corps reservoir data base.

CONCLUSIONS

This paper has demonstrated an analytical approach which provides insights into the adequacy of data for modeling purposes. Potential applications of the approach to monitoring program design are discussed elsewhere (Walker, 1980b). Results suggest that chlorophyll *a* is considerably less sensitive to phosphorus or transparency in these reservoirs, compared with the natural lakes used by Carlson in developing the index system. The phosphorus and transparency indices are not relative indicators of biomass in these reservoirs, possibly because they are influenced by non-algal materials. In future work the approaches demonstrated here will be applied in evaluating alternative schemes for summarizing relationships among measures of trophic state in reservoirs, using an expanded data set.

Table 1. — Results of regression analyses relating station-mean index values.

Equation	Slope standard error	R^2	Mean squared error
standard regressions			
$I_B = 24.7 + .443\ I_P$	.033	.374	37.2
$I_B = 25.2 + .403\ I_T$	.036	.291	42.1
$I_T = 11.6 + .854\ I_P$	.026	.774	24.1
geometric mean regressions			
$I_B = 9.9 + .724\ I_P$	.033	.224	46.1
$I_B = 5.7 + .747\ I_T$	.036	.079	54.7
$I_T = 5.4 + .971\ I_P$	.026	.760	25.6

Table 2. — Results of error analyses.

Relationship[a]	Mean squared error	Data error	Percent data error
I_B/I_P	37.2	21.8	59%
I_B/I_T	42.1	22.0	52%
I_T/I_P	24.1	12.1	50%

[a] Standard regressions in Table 1

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RESERVOIR WATER QUALITY SAMPLING DESIGN

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ABSTRACT

The design of monitoring programs often serves as a major source of error or uncertainty in water quality data bases. Properly designed programs should minimize uncertainty or at least provide a means by which variability can be partitioned into recognizable components. While the design of sampling programs has received recent attention, commonly employed strategies for limnological sampling of lakes may not be completely appropriate for many reservoirs. Reservoirs differ from natural lakes in that they are generally larger, deeper, and morphologically more complex. Reservoirs also receive a majority of the inflow from a single tributary located at considerable distance from the point of outflow. The result is the establishment of marked physical, biological, and chemical gradients from headwater to dam. The existence of horizontal as well as vertical gradients, and their importance in water quality sampling design were the subject of intensive transect sampling efforts at DeGray Lake, a U.S. Army Corps of Engineers reservoir in southern Arkansas. Data collected were used to partition variance, identify areas of similarity, and demonstrate how an equitable sampling program might be designed.

INTRODUCTION

Recent legislation, including the Federal Water Pollution Control Act (P.L. 92-500) and the Amendments of 1972 and 1977, requires water quality monitoring programs to idantify problems and assess management procedures. As a result, Federal, State, and local agencies spend millions of dollars annually monitoring water quality in rivers, lakes, and reservoirs. Often overlooked in the final analysis, however, is the error or uncertainty associated with these estimates. This uncertainty may result from experimental design, sampling variability, analytical error, intrinsic variability, or all of these. In many instances, the sampling program's design is the major source of bias or error in the data. Designs are often inappropriate because of ambiguous objectives, lack of knowledge about the system, or manpower and funding constraints. Monitoring programs must, therefore, receive careful review and consideration prior to their implementation if meaningful information is to be obtained.

Although sampling design and the problem of uncertainty have recently received attention (Kwiatkowski, 1978; Liebetrau, 1979; Reckhow, 1979, 1980; Reckhow and Chapra, 1979; Ward, et al. 1979), water quality sampling design for reservoirs has not been adequately addressed. This is due, in part, to the tacit assumption that lakes and reservoirs are similar. The purpose of this paper is to: (1) Generally describe several differences between reservoirs and lakes that influence sampling design; (2) discuss an intensive water quality sampling program conducted at an U.S. Army Corps of Engineers reservoir; and (3) describe one approach for designing reservoir water quality sampling programs.

LAKES AND RESERVOIRS

Although reservoirs are incorporated in the formal definition of lakes (Hutchinson, 1957), several significant differences between lakes and reservoirs suggest that reservoirs are unique lentic systems (Ryder, 1978; Thornton, et al. 1980). A comparison of 309 natural lakes and 107 USAE reservoirs included in the 1972-75 U.S. Environmental Protection Agency National Eutrophication Survey indicated reservoirs had greater drainage and surface areas, drainage/surface area ratios, mean and maximum depths, shoreline development ratios, and areal water loads than did natural lakes (Table 1). Reservoirs also had shorter hydraulic residence times and lower total phosphorus and chlorophyll concentrations despite higher total phosphorus and nitrogen loadings.

In addition to these differences in scale, reservoirs also exhibited pronounced longitudinal gradients, a phenomenon not unexpected considering the importance of advective and unidirectional transport in reservoirs (Baxter, 1977). Impoundment of meandering rivers and their floodplains often creates long, narrow, highly dendritic reservoirs that receive most of their inflow from a single tributary located a considerable distance from the outflow or dam. This promotes the development of physical, chemical, and biological gradients in space and time (Gloss, et al. 1980; Hamblin and Carmack, 1978; Hebbert, et al. 1979; Hyne, 1978; Johnson and Merritt, 1979; Kennedy, et al. 1980; Kimmel and Lind, 1972; McCullough, 1978;

Table 1. — A comparison of geometric means on selected variables for natural lakes and USAE reservoirs (Thornton, et al. 1980).)

Variable	Natural Lakes (N = 309)	USAE Reservoirs (N = 107)	Probability Means are Equal
Drainage Area (Km^2)	222.0	3228.0	<0.0001
Surface Area (Km^2)	5.6	34.5	<0.0001
Drainage/Surface Area	33.0	93.0	<0.0001
Mean Depth (m)	4.5	6.9	<0.0001
Maximum Depth (m)	10.7	19.8	<0.0001
Shoreline Development Ratio	2.9 (N = 34)[*]	9.0 (N = 179)[**]	<0.001
Areal Water Load (m/yr)	6.5	19.0	<0.0001
Hydraulic Residence Time (yr)	0.74	0.37	<0.0001
Total Phosphorus ($\mu g/l$)	54.0	39.0	0.02
Chlorophyll *a* ($\mu g/l$)	14.0	8.9	<0.0001
P Loading ($g/m^2 \cdot yr$)	0.87	1.7	<0.0001
N Loading ($g/m^2 \cdot yr$)	18.0	28.0	<0.0001

[*] Hutchinson, 1957
[**] Leidy and Jenkins, 1979

Thornton, et al. 1980). These gradients should be considered in designing reservoir water quality sampling programs.

DEGRAY LAKE

Description and characterization of lateral, longitudinal, and vertical water quality gradients were the objectives of intensive water quality transect samplings conducted on DeGray Lake, a USAE reservoir located in southern Arkansas. DeGray Lake has a 53.4 km² surface area, a mean and maximum depth of 9 and 60 m, respectively, and is located in a large (1,162 km²) predominately forested watershed. It is highly dendritic (shoreline development ratio of 13) and exhibits strong thermal stratification. DeGray Lake has an average hydraulic residence time of 1.2 years and is operated for hydropower production. The outlet structure has the capability for selective withdrawal and can discharge epilimnetic, metalimnetic, or hypolimnetic water to meet downstream requirements.

METHODS

Sampling transects were established from the dam to the headwaters. Stations were located on each transect to sample over the old river channel, in the littoral area on each shore, and at intermediate distances between these locations. Fifteen transects, averaging five stations per transect, were established in the main body of DeGray Lake (Figure 1). Three to four transects were also established on the two major embayments. Water samples, pumped to the surface from depths of 0, 2, 4, 6, and 10 m and at 5-m intervals thereafter to within 0.5 m of the bottom, were stored in acid-washed polyethylene bottles. Sampling occurred during July 1978, and January and October 1979. All samples were collected on the same day during each sampling trip. The lake was thermally stratified during the summer and isothermal during the winter. During stratified periods, the lake was divided into epilimnion, metalimnion, and hypolimnion by defining the metalimnion as those depths at which temperature changed by 1°C or more per meter of depth. The lake was completely mixed during the January sampling.

Laboratory determinations included total phosphorus, chlorophyll *a*, and turbidity analyses since these parameters generally characterize the distribution of a representative nutrient, phytoplankton biomass, and physical factors affecting light regime, respectively. Total phosphorus samples were analyzed by persulfate digestion/ascorbic acid reduction according to standard methods (Am. Pub. Health Assoc. 1976). Turbidity analyses were conducted on a Hach turbidimeter. Chlorophyll samples, taken only at 0 and 4 m, were stored in the dark at 4°C in polyethylene bottles, and filtered within 8 hours. The trichromatic method, with pheophytin correction, was used for chlorophyll analyses (Am. Pub. Health Assoc. 1976). Chlorophyll samples that could not be analyzed immediately were frozen after filtering and analyzed within 2 weeks.

Data analyses were performed using the Statistical Analysis System (SAS 79.4, 1979). A variance component analysis with random group effects was used to test for differences with depth, sampling station, and transect. Since samples were not replicated, sampling depth was nested in the analysis of variance and included sampling and analytical error as well as the variability among sampling depths.

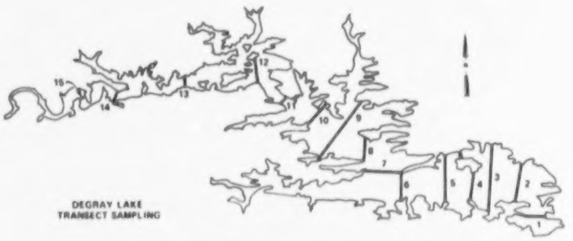

Figure 1. — Map of DeGray Lake, Arkansas, and locations of sampling transects (bold lines).

RESULTS

DeGray Lake exhibited marked longitudinal and vertical variation during all sample trips as evidenced by the percentage of the total variance contributed by

Table 2. — Percent of total variance.

	Epilimnion			Metalimnion			Hypolimnion		
Variable	Tran	Sta	Depth	Tran	Sta	Depth	Tran	Sta	Depth
Turbidity	76	0	24	69	0	31	10	9	81
Total P	55	0	45	56	0	44	33	13	54
Chl *a*	88	0	12						
			OCTOBER						
Turbidity	73	3	24	40	28	32	48	2	51
Total P	18	8	74	61	4	35	98	0	2
Chl *a*	83	0	17						
			JANUARY (no stratification)						
Turbidity	97	0	3						
Total P	60	2	38						
Chl *a*	15	26	60						

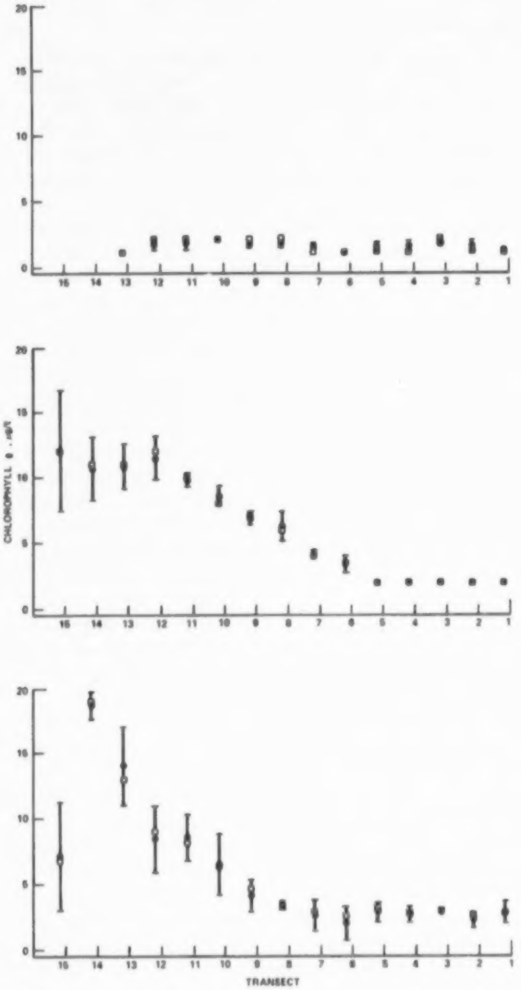

Figure 2. — Changes in mean (solid circle) and median (square) chlorophyll *a* concentration (ug/l)b with transect in January 1978 (top), July 1979 (middle), and October 1979 (bottom). Vertical lines indicate 1 S.D.

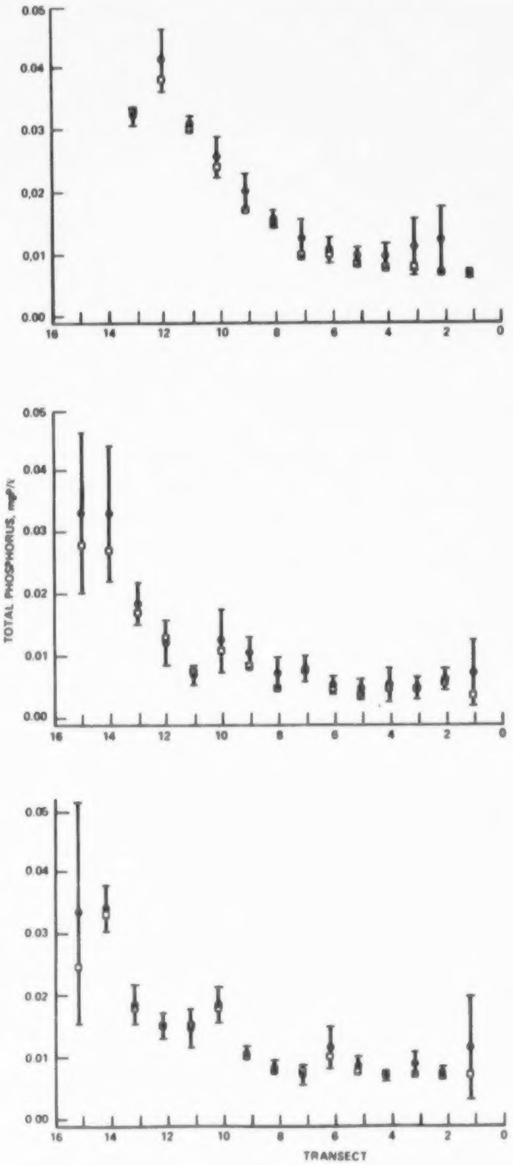

Figure 3. — Changes in mean (solid circle) and median (square) total phosphorous concentration (ug/l)b with transect in January 1978 (top), July 1979 (middle), and October 1979 (bottom). Vertical lines indicate 1 S.D.

transect and depth, respectively (Table 2). Since variance associated with stations within transects was minimal, subsequent comparative analyses were performed using volume-weighted transect means for epilimnion, metalimnion, and hypolimnion.

A comparison of these transect means indicated that longitudinal variation could be attributed to the existence of gradients from headwater to dam. In general, turbidity and total phosphorus concentrations in the epilimnion, metalimnion, and hypolimnion

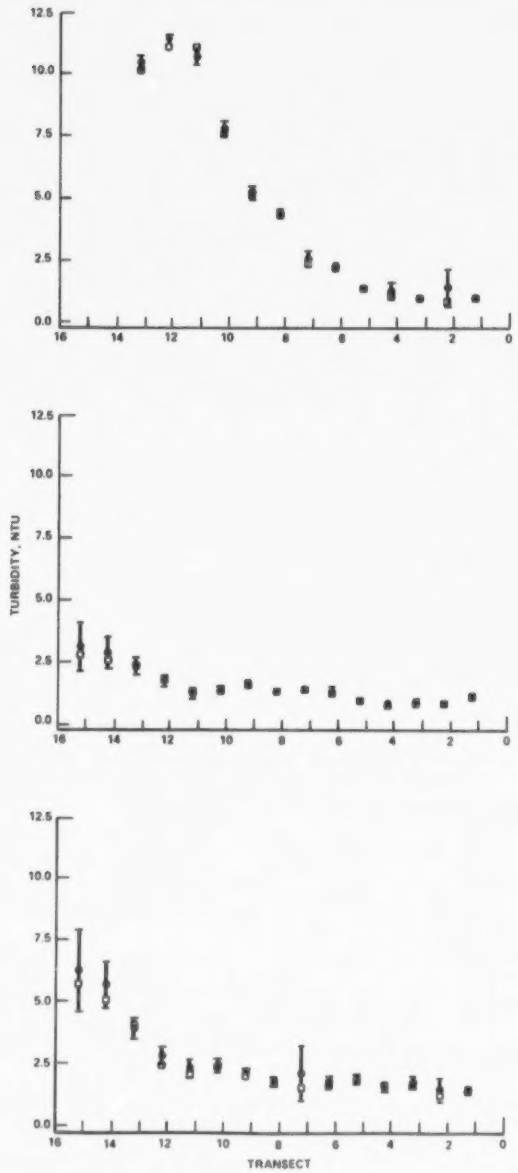

Figure 4. — Changes in mean (solid circle) and median (square) turbidity (NTU's) with transect in January 1978 top), July 1979 (middle), and October 1979 (bottom). Vertical lines indicate 1 S.D.

decreased with distance downstream during all sampling trips. Epilimnetic chlorophyll *a* concentrations, while decreasing in a downstream direction in July and October, were lower and similar for all transects during the January sampling (Figure 2). Of limnological significance is the fact that differences among transects were most pronounced in the upper region of the reservoir nearest the tributary inflow. Differences among transects from mid-reservoir to dam, when they existed, were minimal. For example, mean epilimnetic total phosphorus concentrations for headwater transects ranged from 0.03 to 0.04 mgP/l during all sampling months, decreased to approximately 0.01 mgP/l by transect 6, 11, and 9 in January, July, and October, respectively, and then remained unchanged between these mid-reservoir locations and the dam (Figure 3). Mean epilimnetic turbidity values, which were lowest during the July sampling, exhibited longitudinal changes similar to those for total phosphorus (Figure 4). The variance about transect means was, in general, greatest at upstream transects on all sampling dates and lowest at transects near the dam.

These longitudinal gradients present a unique problem in designing a water quality sampling program. Since it is not possible to sample at a single station and adequately characterize reservoir water quality, the number and location of required sampling stations must be determined. One approach for locating multiple stations will be illustrated using epilimnetic total phosphorus, turbidity, and chlorophyll *a* data from DeGray Lake.

Analyses of variance and Duncan's multiple range test indicated no significant differences among certain transects as well as significant differences among others. Various linear models were used to describe the change in epilimnetic mean total phosphorus and chlorophyll *a* concentrations, and turbidity longitudinally down the reservoir. These linear models provided an initial estimate of the minimum number of stations required to characterize these areas. If, for example, ANOVA procedures are to be used to characterize areas, and the slope (b) of a model among transect means is not significantly different from zero, then a minimum of one station would be required to characterize this area. If a linear function $(b \neq 0)$ accounts for a significant portion of the variance among other transect means, a minimum of two stations might characterize this area; a quadratic function would require three stations; a cubic function, four stations, etc. It should be noted that the suggested numbers of stations are minimums, since increasing the number of stations would provide greater reliability and statistical confidence in all estimates, particularly those described by statistical models. These linear models and appropriate transects can then be compared for all water quality variables of interest and all sampling dates to identify areas of similarity and overlap.

For DeGray Lake, transects 1 through 5 exhibited similar means for all variables over all dates, so a single station could be selected to characterize this area. The relation among transects 6 through 13 (i.e., mid-reservoir) was generally linear, thus requiring a minimum of two sampling stations to characterize water quality gradients in this area. Since transects 14 and 15 were either distinct or linear, a separate station could be established for each transect. A minimum of five sampling stations, then, would be required to characterize longitudinal water quality gradients in DeGray Lake. These might logically be located on transects 3, 10, 12, 14, and 15. Since there was no significant lateral variability, the stations could be located over the deepest point on each transect.

The number of samples to be collected at each station depends on the variability of the water quality constituent and the desired precision of the estimate. Estimates of variability can be obtained by reviewing existing data or, as in the case of DeGray Lake, by preliminary surveys of the system to be sampled. Precision will be dictated by analytical capabilities and/or the purposes for which the data will be used. A general formula for random sampling can be used to obtain initial estimates for sample size (Cochran, 1963). A Student's t value for 30 degrees of freedom can be used to initiate the procedure. The formula is applied iteratively until n converges on the sample size.

$$n = \frac{t^2 s^2}{d^2}$$

where n = number of samples
t = appropriate value from Student's t distribution
s^2 = sample variance
d = desired precision about the mean

Assuming a fixed cost for a sampling trip, the cost of sampling several variables can be estimated by

$$C(n) = C_o + \sum_{i=1}^{k} C_i n_i$$

where C_o = fixed cost of sampling
C_i = unit sample cost for variable i
n_i = number of samples for variable i

Frequently, the total computed sampling cost will exceed the funds available and the sampling effort must be reduced. Since precision is incorporated in the sampling formula, a matrix can be developed to indicate concomitant reductions in cost and precision of various water quality constituents (Table 3). For example, we may wish to estimate means for total phosphorus at the five suggested sampling stations in DeGray Lake within 5 μgP/l with 95 percent confidence. Since means for these stations differ, percent precision will vary from 50 percent at transect 3 (mean of 9 μgP/l) to 15 percent at transect 14 and 15 (mean of 34 μgP/l) . Finding the appropriate entry in Table 3 indicates that 12 samples will be required at transect (or station) 3, 18 samples each at transects 10 and 12, and 23 samples each at transects 14 and 15. The total number of samples for all stations would thus be 94. Assuming a unit analytical cost of $13, the total analytical cost per sample trip would be $1,222. If we would also like to estimate means for chlorophyll within 25, 25, and 15 percent, and turbidity within 50, 20, and 20 percent at downstream, mid-reservoir, and headwater stations, respectively, with 95 percent confidence, analytical cost could then be obtained by summing the cost for all three variables. In this example, the total analytical cost per sampling trip would be $3,085. Assuming a fixed sample collection cost of $412 per sample trip, the total cost of this three-variable sample program would be $3,497 per sample trip.

If it is determined that funds would be insufficient to support this sampling effort, then some decision concerning the quality of the data would have to be made. For instance, if the objectives of the study required precise information for total phosphorus (e.g., ± 5μgP/l but less precise information for chlorophyll and turbidity (i.e., >25 , 25, and 15 percent and >50 , 20, and 20 percent, respectively), then the number of samples for chlorophyll and turbidity could be reduced.

Table 3. — Decision matrix for DeGray Lake epilimnetic sampling.

		Transect 3						Transect 10 or 12						Transect 14 or 15					
	Unit Cost	$13						$13						$13					
	Mean	9 μgP/l						20 μgP/l						34 μgP/l					
Total	Precision	±50%			±100%			±25%			±50%			±15%			±30%		
Phosphorus	Probability	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%
	Sample No.	12	9	5	4	4	3	18	13	8	6	5	3	32	23	14	10	7	5
	Total Cost	156	117	65	52	52	39	234	169	104	78	65	39	416	299	182	130	91	65
	Unit Cost	$3						$3						$3					
	Mean	1.3 NTU's						5.3 NTU's						5.3 NTU's					
Turbidity	Precision	±50%			±100%			±20%			±40%			±20%			±40%		
	Probability	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%
	Sample No.	13	10	6	5	4	3	8	6	4	4	2	1	8	6	4	4	2	1
	Total Cost	39	39	18	15	12	9	24	18	12	12	6	3	24	18	12	12	6	3
	Unit Cost	$20						$20						$20					
	Mean	2 μg/l						7 μg/l						11 μg/l					
Chlorophyll	Precision	±25%			±50%			±25%			±50%			±15%			±30%		
	Probability	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%	95%	90%	80%
	Sample No.	19	13	9	6	5	3	13	10	6	5	4	3	21	15	9	7	5	4
	Total Cost	380	260	180	120	100	60	260	200	120	100	80	60	420	300	180	140	100	80

NOTE: 1. Mean values are those expected based on three sampling dates.
2. Percent precision based on approximate levels of analytical precision for each test or requirements of the study.
3. Total cost calculated as product of unit cost and sample number.

This, in turn, would reduce analytical costs. For example, decreasing the percent precision for chlorophyll and turbidity by a factor of two while continuing to estimate total phosphorus within ±5 μgP/l would cost $1,885. Assuming 12 sample trips per year, this would save $14,000 per year.

Construction of a similar matrix for all water quality variables, including information for the metalimnion and hypolimnion, would allow design of a total sampling program. This matrix permits balancing precision, confidence, and cost within monetary constraints. While loss of precision may be scientifically undesirable, at least the uncertainty associated with the data could be accounted for in comparing and discussing the data.

Finally, sampling dates, which will depend in part on the objectives of the sampling program and site-specific characteristics, must be identified. Since reservoirs are strongly influenced by advective transport of nutrients and particulate material, sampling during elevated flows is necessary. Sampling should also occur during periods of stratification and during low flow periods. One potential sampling strategy for DeGray Lake might be: Once during January; three times during elevated flows from March to mid-April; once during May; three times during stratification from mid-June through July; once in August; three times during low flow from mid-September to mid-October; and once in mid-November. The total number of samples approximates a monthly sampling effort but the sampling program now incorporates hydrologic and limnological factors responsible for many of the observed water quality gradients. Obviously, the exact dates must reflect the objectives of the sampling program, potential problems, and specific hydrologic and site-specific characteristics. In temperate regions, spring runoff is a critical period that needs to be incorporated in the reservoir sampling program. Winter kill in some reservoirs may require more frequent winter sampling.

DISCUSSION

Minimizing uncertainty or partitioning variability into recognizable components is the purpose of experimental design. To be effective, sample design should allow the researcher to adequately discuss the characteristics of the system as well as permit comparative evaluations through time and/or across systems. The presence of non-partitioned variability in the data reduces its informational content and thus its value to the researcher or manager.

Proper design of a sampling program will depend in part on the objectives of the study. An objective common to many limnological surveys is characterization of the water quality of an entire body of water. However, it is frequently assumed that horizontal heterogeneties are insignificant relative to those occurring vertically, and thus a single, deep station is often established. In lakes such as DeGray, which exhibit marked longitudinal gradients, such an approach to sample design would be inappropriate. For instance, using Vollenweider's (1968) criteria to classify lakes with respect to phosphorus and chlorophyll concentrations, DeGray Lake could be classified as either oligotrophic, mesotrophic, or eutrophic depending on the location of the single sampling station. On all sampling dates, the headwater area would be classified as eutrophic, mid-reservoir as mesotrophic, and the lower reservoir as oligotrophic. DeGray Lake is not unique in this regard, as similar classification problems have arisen in other reservoirs (Hannan, et al. 1980).

The approach to sample design outlined here for DeGray Lake provides a means for obtaining information which can improve characterization of the entire lake as well as the existence of gradients which may be of limnological or management significance. Characterization of the entire lake, if it must be attempted, could be more realistically accomplished by volume-weighting observations from representative portions of the lake, while station-by-station evaluation of the same information would assess possible problem areas.

An additional consideration in the design of a sampling program is to balance desired precision against the reality of monetary limitations. A review of historical data, educated guesses, or, as in the case of DeGray Lake, data from a pilot study will be required to estimate initial means and associated variability, and thus, initial estimates of sample size (Cochran, 1963). For long-term monitoring programs, the expenses of a pilot study may be more than offset by eliminating samples which do not significantly increase the informational content of the data base. Such studies would also identify the existence of gradients, information which may be important in meeting study objectives. Identification of the general location of these gradients prior to beginning a monitoring program would permit logical positioning of sampling stations.

CONCLUSIONS

Reservoirs are advectively dominated systems that often exhibit pronounced gradients in water quality. Evaluation of water quality conditions in reservoirs will, therefore, require sampling programs designed to adequately assess changes in both space and time. Presented here is one approach for designing such a program, using DeGray Lake as an example. Similar studies currently being conducted at other reservoirs will allow further evaluation of this approach and hopefully provide a basis for generalizing sampling design methodologies for use in many reservoirs.

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HEALTH ASPECTS OF EUTROPHICATION

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ABSTRACT

Increasing eutrophication makes water more difficult and expensive to treat. Soluble organic matter derived from algae contributes to taste and color and produces chloroform and other trihalomethanes when the water is chlorinated. These compounds are suspected of being weakly carcinogenic for humans and are indicators of the formation of other chlorinated compounds that have been less well studied. The adverse effects directly related to eutrophication are, in general, diseases transmitted by vectors, such as mosquitos and snails which spend at least part of their life in water. Different mosquito species have different water requirements. Yet, many water-related health problems exist independently of eutrophication. Waterborne bacteria and viruses, as well as protozoa and other parasites that do not have an intermediate host, are simply carried by water from an infected person to susceptible persons and eutrophication has no direct influence on the passage. It is not possible to state that increased eutrophication will affect the incidence of mosquito-borne diseases. Snails are quite adaptable and reducing eutrophication is not likely to be an effective method for controlling schistosomiasis. Direct health effects of algae produced in eutrophied lakes include objectionable taste and odor, dermatitis, asthma, other allergic responses, and low grade chronic toxicity. Few cases of human poisoning by algae have been reported, probably because the associated taste and odor causes people to seek other sources of water whenever possible.

INTRODUCTION

The voluminous literature on eutrophication contains remarkably few references to health effects. To find such information one must search under "artificial or manmade lakes," "sewage lagoons," or "oxidation ponds." One of the most comprehensive treatises on the subject is the book "Man-made Lakes and Human Health," a collection of 31 papers (Stanley and Alpers, 1975).

Most manmade lakes pass through an initial first-fill eutrophic stage produced by nutrients leached from the soil or decaying vegetation. If there are no major nutrient inputs to the lake, this stage may be transitory as nutrients are flushed out or buried in sediment. Sewage lagoons are heavily loaded with nutrients which produce a high state of eutrophication at the "clean" end of a series.

Many water-related health problems exist independently of the state of eutrophication. Waterborne bacteria and viruses, as well as protozoa and other parasites that do not have an intermediate host, are simply carried by water from an infected person to susceptible persons and eutrophication has no direct influence on the passage.

While a high degree of eutrophication is universally bad, a moderate degree may have some beneficial effects. Increasing eutrophication makes water treatment more difficult and more expensive. An example of this problem, which is classical in its development, is the Bou Regreg water supply reservoir in the Atlas mountains above Rabat in Morocco. The treatment plant has been found inadequate because of the serious taste and odor problem caused by algae. Treatment with large quantities of activated carbon now appears to be necessary to produce a water that will have an acceptable taste (WHO). If the water is not acceptable, there will be a serious health risk because consumers will turn to other, possibly unsafe drinking water sources (Eng. Sci. Med. 1979).

The adverse effects directly related to eutrophication are, however, diseases transmitted by vectors, such as mosquitos and snails which spend at least part of their life cycle in water. Of the very large number of species, only a few can carry disease organisms that infect man, while others are a general nuisance, their bites causing allergic reactions and sometimes secondary infection.

MOSQUITOS

In the case of malaria, early on it was discovered that it was important to concentrate on the species that carried the disease in the specific area in question (Waddy, 1975; Surtees, 1975). Different mosquito species have different water requirements: Some hatch from clear water in sunlight, others hatch from shaded pools of water held by plant leaves, and still others flourish in mats of algae on eutrophied lakes. Therefore, it is not possible to state that increased eutrophication will increase or decrease the incidence of mosquito-borne diseases. In many cases, the mosquitos hatch from small pools of water adjacent to the main body of the lake and only few, if any, can survive in open water where they are subject to predation by fish and disturbance by waves.

Underwater rooted vegetation and heavy mats of algae or floating water plants increase the number of sites where mosquitos can hatch and, therefore, may increase the incidence of arthropod-carried disease, including malaria, dengue fever, disease due to arboviruses and filariasis (Waddy, 1975).

One objection to sewage treatment lagoons, which certainly are highly eutrophied, is that they provide breeding places for *Culex* mosquitos, the primary vectors of several types of encephalitis virus. In the midwestern and southwestern United States, improperly maintained lagoons were found to be a major source of these mosquitos (Hopkins, 1960). Control was achieved by removing rooted vegetation and algae mats. Fish that eat mosquito larvae are frequently cultivated in lakes, ponds, and rice paddies (Jackson, 1975). These mosquito-fish are too small to contribute very much to human nutrition, so their only effect is to control the mosquito population. High levels of eutrophication produce shelter for mosquito larvae where fish cannot reach them. Many mosquitos do not require a high level of nutrients and the Arctic, which is famous for numerous clear oligotrophic lakes, still swarms with myriads of mosquitos and small flies during the summer.

Nutrients from sewage discharge in the Baltic Sea have been blamed for increased growth of the *Phragmites* reed in shallow water near coastal cities. These dense growths probably increase the number of mosquitos which, in this climate, are not serious vectors of disease but certainly interfere with man's well-being.

SNAILS

In the tropics, schistosomiasis (bilharziasis), a debilitating but rarely fatal disease caused by a trematode worm, seems invariably to accompany the construction of lakes and irrigation schemes (Jordan, 1975 ; Burch, 1975). The schistosome spends half of its life cycle in a freshwater snail. Infected humans excrete worm eggs, larva hatch from these eggs and can develop in certain species of snails if excreta are discharged into standing or slow moving water. The developed infectious larvae leave the snail after a few days and enter the skin of humans who may be wading in the water. Theoretically, this chain of infection can be broken by good sanitation, keeping excreta (including urine) out of surface water, controlling snails, chemotherapy of infected persons, and keeping humans, especially children, out of the water. The snails that harbor the schistosomes require a tropical climate and this is, unfortunately, the very climate that encourages children to play, bare-legged, in the water. The host snails thrive in shallow waters that contain dissolved organic matter and are mildly eutrophic. They grow best when the water has moderate light penetration and is not turbid. An abundance of submerged aquatic plants provides shelter for egg laying. However, the host snails are quite adaptable and reducing eutrophication is not likely to effectively control schistosomiasis.

ALGAE

Direct effects of algae produced in eutrophied lakes include objectionable taste and odor. Dermatitis, asthma, and other allergic responses, and low grade chronic toxicity have also been reported in a few cases (MacGregor and Keeney, 1975). Diarrhea has been demonstrated in experimental animals, and livestock have been killed by drinking water that was heavily infected with blue-green algae (cyanophytes). In the latter case, the livestock had no choice but to drink the algae with the water. However, it is uncertain whether the toxic material was in solution or in the algae cells. On the other hand, only a few cases of human poisoning by algae have been reported, probably because the associated taste and odor have caused people to seek other sources of water whenever possible (Kay, Sykora, and Burgess, 1980). Eutrophication in salt water may be manifested as a red tide of dinoflagellates. These small organisms are filtered out by shellfish, rendering them toxic. The prevalence of dinoflagellates in the warmer summer months is probably the origin of the old stricture against eating shellfish in any month without an R in it (May-August).

Soluble organic matter derived from algae contributes taste, odor, and color to water but also produces chloroform and other trihalomethanes when the water is chlorinated. These compounds are suspected of being weak carcinogens for humans and are indicators of the formation of other chlorinated compounds that have been less well studied (WHO, 1978).

The secondary effects of high levels of eutrophication, though not directly affecting health, nevertheless detract from man's well-being. A high concentration of algae in bloom causes extreme shifts in dissolved oxygen and pH, and may kill fish and other, less tolerant, species of algae. Even the algal species responsible for the bloom will eventually die of overcrowding. Bacterial decay of the dead plant and animal matter removes dissolved oxygen and then generates hydrogen sulfide which further contributes to objectionable odors and foul black deposits of ferrous sulfide. Aquatic weeds can completely clog lakes and waterways that receive excess nutrients, rendering them unfit for navigation, fishing, or even as a water supply (Ferguson, 1968).

OTHER EFFECTS

Eutrophication may have an indirect and possibly beneficial effect on the transmission of waterborne diseases. There is evidence that algae and plants are antagonistic to enteric bacteria so that the effluent of a well-run set of oxidation ponds has a lower count of indicator organisms than the effluent from a trickling filter or activated sludge plant (Hopkins, 1960). A less direct effect of algae in a water supply is to increase the probability that the water will be treated prior to consumption. Treatment that reduces turbidity also removes many infective agents. Man is all too willing to drink polluted water if it looks and tastes fairly good and eutrophication may serve as a warning that treatment should be applied.

The construction of large artificial lakes displaces the local population, causing interruptions in food supplies, dependence on government hand-outs, and psychological disturbances. If the lake supports a suitable population of fish, they can make a very real contribution to the nutrition and well being of the displaced population (Hopkins, 1960). First-fill eutrophication usually provides excellent fishing in the first few years of a reservoir's life. If no nutrients are added, the fish catch will eventually decline as the original nutrients are depleted. Good fishing requires a controlled state of eutrophication. Even clean-water fish, such as salmon, may be more productive if controlled amounts of fertilizer are added to the water (Le Brasseur, McAllister, and Parsons, 1979).

WHO ACTIVITIES

A number of WHO activities have been concerned, directly or indirectly, with the health aspects of eutrophication (Deom, 1976; Landner, 1976). Malaria and other parasitic diseases have a high priority and are handled through specific programs by both WHO headquarters and the regional offices. Moreover, the World Health Organization cooperates in the international effort to provide safe drinking water and sanitation for all by the end of this decade. To achieve this goal, pollution from sewage and agriculture must be reduced; one of the results will be a decrease in the eutrophication of lakes and slow moving rivers.

CONCLUSION

The effects of eutrophication on man's well-being are not very extensive and they may be only partially negative at low levels of eutrophication. A high level of eutrophication is generally bad, but even here the effects on disease are only indirect, and there seems to be no justification at present for special programs dealing with the effects of eutrophication on disease. Present programs to control excessive eutrophication will, to the extent that they are successful, have a favorable effect on human disease as well as make a positive contribution to the health of the total environment.

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GENERAL IMPACTS OF EUTROPHICATION ON POTABLE WATER PREPARATION

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ABSTRACT

With increasing eutrophication, the preparation of potable water from lakes and reservoirs is so disturbed by algal blooms and the products of algal metabolism and decay, that the security of potable water supply at times becomes questionable. Particularly the following disturbing factors can be enumerated: Penetration of algae through the treatment plant into potable water, inducing aftergrowth of bacteria, caused by the decay of algal substances in distribution systems, reservoirs, and end points; dissolved organic substances originating from algae consuming chlorine impair disinfection of potable water; if chlorine is used for oxidation and disinfection these compounds are precursors for the formation of trihalomethanes; considerable algal counts in raw water clog filters; certain algae release taste and odor substances that cause taste and odor impacts on water. Eutrophication of water depletes the oxygen concentration and leads to the reductory release and increasing concentration of iron- and mangan-ions in the hypolimnion. Mangan-II-ions are often difficult to remove from water. Frequently eutrophic water contains increased concentrations of ammonia which disturb the disinfection process. Drinking water reservoirs should be kept in an oligotrophic, eventually mesotrophic, but by no means in an eutrophic status.

INTRODUCTION

Experience in Germany with treating water from reservoirs for drinking water during the last decades has shown that the main disturbance in the treatment process was caused chiefly by eutrophication of the impounded water. The following influential factors have negative effects on the quality of the impounded water:

1. Domestic, sometimes also industrial sewage.
2. Effluents from farms and cattle-breeding.
3. Effluents from cultivated land (erosion, flushing).
4. Natural phenomena, e.g., moor water, decaying leaves, humates, mine drainage (e.g., lead, cadmium, zinc).
5. Biocides (used in agriculture and forestry).
6. Dangerous organic substances, especially those that are persistent.

This paper is concerned only with the effects of eutrophication on stagnant waters used as drinking water.

LOADING WITH PHOSPHORUS AND NITROGEN COMPOUNDS

From the point of view of usage, the eutrophication of a stagnant water body represents an undesirable change in the quality of the water. It is the constant inflow of too many phosphorus and nitrogen compounds which is particularly responsible for this negative quality of the water. Phosphorus is of chief importance because it acts as a limiting factor in most stagnant water bodies and determines the extent of phytoplankton production.

If the average annual P-concentration in the tributaries exceeds the specific concentration limit tolerable for a given stagnant water then plankton production is likely to increase to such an extent that more biomass is produced in the lake than can be decomposed under aerobic conditions on the bottom.

An input of nitrogen compounds does not influence primary production if phosphorus is the limiting factor. Despite this, one should attach a certain amount of importance to an inflow of nitrate ions, nitrite ions and ammonium ions because they can either disturb the water treatment process or impair the quality of the drinking water.

DETRIMENTAL EFFECTS OF EUTROPHICATION ON WATER QUALITY IN LAKES AND RESERVOIRS

The manmade eutrophication of lakes and especially of drinking water reservoirs causes profound changes, primarily detrimental, in the quality of the water. These detrimental effects impair the process of obtaining drinking water from lakes and reservoirs.

Primary detrimental effects of algal development:

1. Change in algal population as green and blue-green algae increase. The growth of blue-green algae is especially detrimental.
2. Extensive occurrence of particulate organic substances (phytoplankton, zooplankton, bacteria, fungi, detritus).
3. Occurrence of dissolved organic compounds which impart odors and tastes. They are released by the plankton and other microorganisms as products of metabolism or cellular decomposition.

4. Formation of organic compounds with chelating or complexing properties.

5. Formation of humic substances during the decomposition of organisms.

6. Occurrence of water colored by plant pigments.

Secondary detrimental effects of algal mass development:

1. The oxygen budget of the water body becomes highly overstrained by the decomposition of biogenic organic substances. This creates water zones free of oxygen expecially in the sediment-water contact area and above it.

2. Incomplete mineralization of organic substances and release of methane. The sediment and the water near the sediment become enriched with non-mineralized organic substances.

3. Reductive release of iron—and manganese ions from the sediment and subsequent increase in their concentrations in the water.

4. Reduction of nitrate to nitrogen and sulfate to hydrogen sulfide. Increasing concentrations of ammonium ions.

IMPAIRMENT OF PRODUCTION AND DISTRIBUTION OF DRINKING WATER FROM EUTROPHIC RESERVOIRS SHOWN BY EXAMPLES

Following is an overview of the effects which a decrease in the quality of the water of eutrophic reservoirs has on the treatment process of this water and on the distribution of this drinking water to the consumer. This report is based on the Wahnbach Reservoir Association's experience with treating water taken from their reservoir which was eutrophic. Experience with other eutrophic reservoirs was similar. Owing to the limited space of this publication an overview of literature on this subject cannot be given.

Excessive Algal Development

Excessive algal development can impair the treatment process to a considerable extent. During the summer, algae are frequently limited to the upper layers of the lake which means that the raw water taken from the hypolimnion is normally low in algae. However, during the periods of partial and full circulation, algae reach all the depths of the lake and thus also the raw water intake zone.

Insufficient elimination of algae using flocculation and filtration

Over a period of some 10 years the blue-green algae *Oscillatoria rubescens* grew in very large quantities in the Wahnbach Reservoir. Figure 1 shows the depths to which it spread during 1969. Some years during spring and autumn the reservoir water turned red. Large algal accumulations on the surface of the reservoir were not exactly aesthetically beautiful and not a particularly good advertisement for a drinking water reservoir.

There were always severe difficulties during the treatment process in the winter and autumn when the algal filaments in the raw water (up to 300 filaments/ml) broke through the sand filter and reached the water supply system.

Algal elimination by means of flocculation using alum would only have been possible if 200 mg/l aluminum sulfate and 100 mg/l natrium carbonate had been added; this is impossible in practice. The Wahnbach Reservoir Association developed a treatment process which was reasonably satisfactory. It entails adding a double dose of alum with a maximum of 20 mg/1 aluminum sulfate and a single dose of 1 to 2 mg/1 of anionic flocculant aid. Despite this new process, it was impossible to prevent the *Oscillatoria* filaments from breaking through the filter from time to time and thus reaching the drinking water.

Special treatment is required to eliminate the algae from the water. Some species are more difficult to eliminate than others. For example, only 90 percent of the relatively large species *Oscillatoria rubescens* can be removed by using several flocculation processes in succession. The highest elimination rate using flocculation and filtration, especially for diatoms, is 99 percent, or in the case of treatment plants which operate exceedingly efficiently it is 99.9 percent. If there are mass algal developments in a water body, these elimination rates are insufficient and the particulate organic substances cannot be reduced to small amounts. They form deposits in the distribution system and storage tanks and act as a basis for the growth of bacteria, macro-zoo benthos, water-lice, mussels, etc.

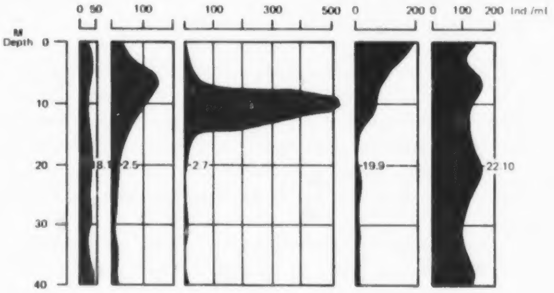

Figure 1. — Distribution of the blue-green algae *Oscillatoria rubescens* in the Wahnbach Reservoir in 1969.

Fitler clogging caused by algae

Mass populations of algae, particularly the large diatoms, rapidly clog a filter and can thus bring operations of a water treatment plant nearly to a standstill. This applies to rapid and slow sand filters. Because these mass algal blooms occur at relatively short notice, little can be done to combat them. This means considerable investment and maintenance expenditure and there is a limit to even these treatment processes. Filamentous algae cannot be retained in a microstrainer satisfactorily.

For example, in the Wahnbach Reservoir the diatom *Melosira islandica* developed in large quantities (approximate chlorophyll concentration of 25 mg/m^3) form Novembver 1972 to January 1973. As a result the filter run time was reduced to 8 hours. At the same time, the blue-green algae, *Coelosphaerium naegelianum*, which form colonies, appeared in increasing quantities. Many of the colonies usually disintegrated into individual cells. The small cells (up to 1,000/ml) could only be eliminated in an unsatisfactory way despite a double dose of flocculant and an additional anionic flocculant aid.

However, it was just this additional anionic flocculant which encouraged the rapid clogging of the filter owing to the presence of diatoms so that eventually filter run times were reduced to 4 hours. Technically, such short filter runs are no longer worthwhile. The water throughput in the plant had to be reduced by 30 percent for a period of 5 weeks until the population of diatoms suddenly broke down in the reservoir at the end of January.

This was the first time that the mass development of various nuisance species of algae disturbed treatment operations to such an extent that the plant throughput was limited. This occurrence must be considered very seriously to demonstrate the fact that development of algae in a stagnant water body can impair the production of drinking water and can even stop it altogether.

Disturbances Caused by Taste and Odor Substances

The occurrence of taste and odor substances in raw water often has a detrimental effect on the drinking water (taste of cucumber, fish, cod-liver oil, earth, etc.). Actinomycetes which often grow following a mass occurrence of blue-green algae are particularly disagreeable and the substances released by these microorganisms impart taste and odor. For example, the organic substance 'geosmine' was isolated from actinomycete populations and blue-green algae. Even when it is highly diluted geosmine still imparts an intense earthy smell.

The mass development of the diatom *Melosira italica* which appeared in the Wahnbach Reservoir during winter circulation in concentrations of up to 100,000 cells/ml imparted a fishy, oily smell and taste to the water; this originated from products of metabolism and decomposition. Specific examinations showed that trimethylamine was present in the water. Even adding activated carbon to the water failed to eliminate the impaired taste and smell. What was particularly unpleasant was the fact that these organoleptically effective substances remained in the filters for weeks after a mass diatom bloom in the reservoir, and they reached the filtrate which meant that the odor and taste of the water was impaired for over 4 weeks. We received numerous complaints from the population during this period.

Disturbances Caused by Dissolved Organic Compounds

One consequence of extensive algal growth in a reservoir is the occurrence of dissolved organic compounds as the products of algal metabolism and decompositon. Even low concentrations of these dissolved organic substances (i.e., between 0.5 to 2 mg/1 DOC) can disturb the water treatment process.

Flocculation Disturbance

Flocculation using iron- and aluminum salts is occasionally disturbed by organic substances of this type. These products are of different molecular weights (Figure 2). Apart from the products of algal metabolism, other products of cellular decomposition are probably also of significance. They are the products of decomposition after decay, and the products of the metabolism of the zooplankton which arise in connection with their feeding on phytoplankton. All these substances undergo further decomposition owing to the bacteria in the water. Some of them are completely mineralized; others are converted into high molecular compounds, e.g, humates which decompose with difficulty. One assumes that some of them are acid polysaccharides. Several of these substances disturb the flocculation process because they (a) are compounds which have a complexing effect; (b) are compounds with strongly acid groups which enrich the negative surface charge of the colloids and suspensoids which are already negatively charged; (c) form insoluble compounds with the iron-III-and aluminum ions added for flocculation.

All these processes hinder or prevent the hydrolytic formation of the polynuclear hydroxcomplexes of the iron-III or aluminum ions, respectively, which are required for destabilizing the negative colloids and suspensoids. These processes can also impair colloid discharging.

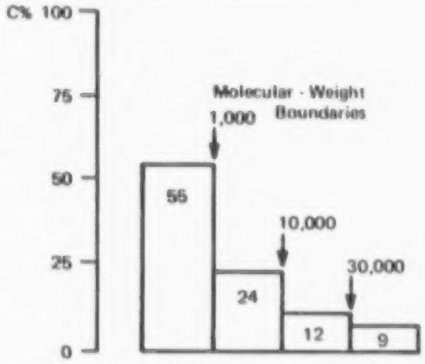

Figure 2. — Percentual proportions of the four fractions of molecular weights of the DOC in the water of the pre-reservoir (November 1979).

Figure 3 shows an example of flocculation disturbed by algal organic substances in the filtrate of an *Oscillatoria rubescens* suspension taken from the Wahnbach Reservoir during a mass algal development. The filtrate contained 14 mg/1 dissolved organic carbon. 10 mg/1 bentonite was added as turbid material for flocculation tests. Experiments showed that it was only after a dose of some 300 mg/1 aluminum sulfate that the negatively charged particles discharged; a dose of 500 mg/1 was required before the particles re-charged. If one compares the course of electrophoretical motion of the bentonite suspension using reservoir water free of algae, it is easy to see that the bentonite particles were already discharged after a dose of 20 mg/1 aluminum sulfate and they become positively charged after 40 mg/1 aluminum sulfate. It was possible to decrease the remaining turbidity only after adding about 300 mg/1 aluminum sulfate. Normally, less than 10 mg/1 aluminum sulfate are sufficient. Flocculation using iron or aluminum salts is disturbed when algal-borne organic substance is present in concentrations of 3 mg/1 dissolved organic carbon. However, during mass blooms of *Oscillatoria rubescens,* we registered algal organic substances in the Wahnbach Reservoir in concentrations of up to 50 mg/1 dissolved organic carbon.

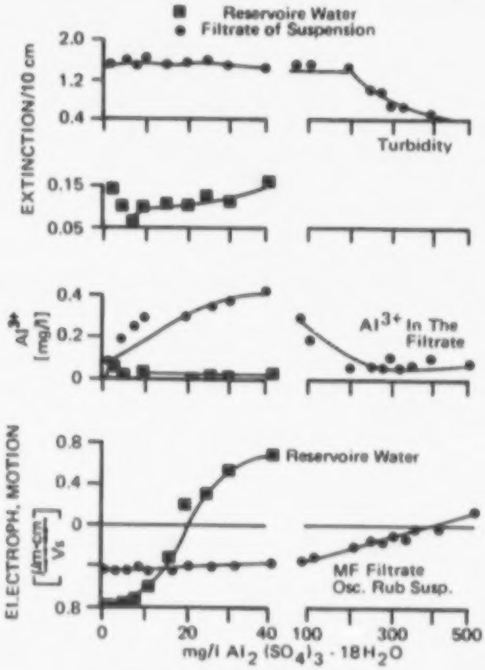

Figure 3. — Disturbance of the flocculation process caused by mass development of *Oscillatoria rubescens* in the reservoir. Results of Jar-tests with the filtrate of the algal suspension.

Disturbance of the Disinfection Process

Increased concentrations of dissolved organic compounds in the water of eutrophic lakes and reservoirs have an unfavorable effect on disinfection using usual methods. Specific examinations showed that the process of germ extinction using chlorine in water containing organic compounds is much slower and sometimes incomplete. The presence of these substances not only causes chlorine depletion or consumption of chlorine dioxide if chlorine or chlorine dioxide are used for disinfecting, but decreases the speed of the disinfection process. Although chlorine dioxide dose not act like chlorine with these organic substances, it is reduced by organic compounds just like chlorine and chlorite ion forms which are harmful to humans. Not more than 0.5 mg/1 chlorine dioxide should be added to the water to prevent the chlorite ion concentration in the water from exceeding this figure. If organic substances are present in a colloid form, they can form a protective coating around the microorganisms, and the disinfectant has difficulty penetrating this coating. Under these conditions far more disinfectant has to be used than is required under normal water treatment conditions.

On the whole, disinfection is impaired in the presence of a dissolved or colloidal organic algal substance. Therefore, these substances decrease the total safety of the drinking water supply. On the other hand, a higher dose of, for example, chlorine as a disinfectant, impairs the taste and odor of the water and creates undesirable toxic substances such as trihalomethane, or chlorite ions when chlorine dioxide is the disinfectant.

Algal Organic Substances as Precursors for the Formation of Trihalomethanes

In treatment plants which take water from eutrophic water bodies, it may be necessary to use prechlorination to exclude problems during coagulation and filtration and to guarantee terminal disinfection with chlorine. However, this measure leads to the formation of trihalomethanes. It is a well-known fact that humic acids react with chlorine forming trihalomethanes. We were also able to confirm the fact that algal organic substances act as precursors (Figure 4). This figure shows the trihalomethane concentration plotted against the reaction time of the chlorine added to the water (chlorine depletion time) of unfiltered algal cultures (thick line) and to the water of the prereservoir during algal bloom (300,000 ind./ml) (dotted line). In these tests the Cl_2: C was kept at 3. These graphs show the similarity of the amount of halomethanes produced at the same reaction time of chlorine with either the organic algal substances in these cultural solutions (unfiltered) or with prereservoir water.

Figure 5 shows the linear connection between the concentration of TOCl and THM compounds after 20 hours of chlorine depletion and the content of dissolved organic substance (precursor) expressed as the spectral absorption coefficient at 280 nanometers (water samples were taken from the eutrophic prereservoir and from the mesotrophic main reservoir). This figure shows that under constant chlorine depletion conditions the concentrations of formed TOCl-compounds and the THM-compounds are proportional to the content of organic substances in these water samples. Some of these organic substances act as precursors for the formation of THM-compounds.

In the Federal Republic of Germany trihalomethane concentrations of 25 μg/l as an annual average are considered permissible but measures should be taken to reduce the trihalomethane concentrations in drinking water to far below this figure. The best way of doing this is to keep the presence of precursors in the raw water as low as possible. This is achieved mainly by controlling the extent of algal development.

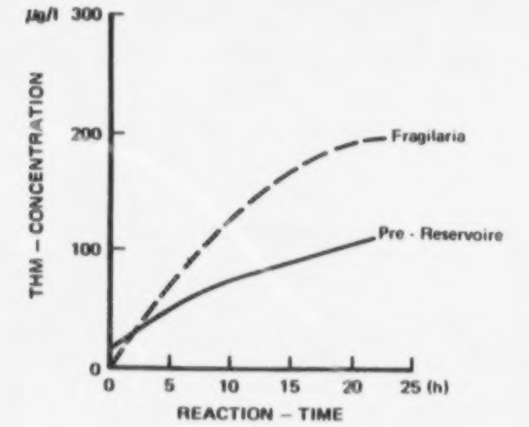

Figure 4. — Formation of THM as a result of the reaction of chlorine with algal organic substances compared with pre-reservoir water rich in algae.

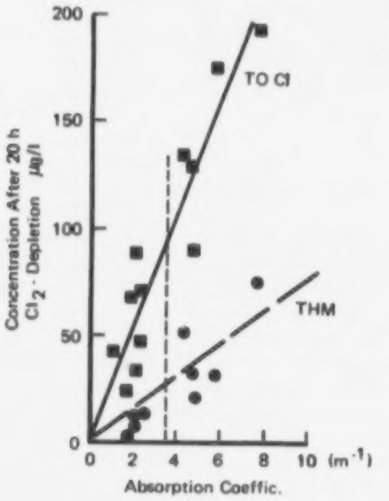

Figure 5. — Formation of the TOC1 and THM depending on the DOC in the water (Absorption coefficient at 280 nm, Cl_2-reaction time 20 h).

The Overstraining of the Oxygen Budget of a Stagnant Water-Body

If the mass of organic substance formed in a stagnant water body during the productivity period exceeds the lake's capacity to decompose this substance under aerobic conditions in the tropholytic zone, then oxygen is depleted and finally disappears in the hypolimnic water, especially on the bottom of the reservoir. Anaerobic conditions lead to the reductive mobilization of various compounds.

Iron and Manganese

During their oxidation, the algal organic substances consume most of the oxygen dissolved in the water and then reduce the manganese-IV-oxide hydroxide hydrates. This increases concentrations of manganese-II-ions in the water, a strong indication of oxygen depletion and of the development of a reduction zone on the bottom of the lake (see Figure 6). Later, the iron oxide hydroxides are reduced and iron-II-ions are released. For example, we have had concentrations of manganese-ions up to 10 mg/1 during summer stagnation in our reservoir after the oxygen had been used up.

Drinking water should contain only extremely small concentrations of iron and manganese compounds. This means extensive iron and manganese elimination in those drinking water plants in which large quantities of manganese and iron are present in the raw water can be a difficult process from a technical point of view.

FUTURE PROSPECTS

These examples demonstrate to what degree the eutrophication of a stagnant water body mainly used for drinking water supply can impair the production and distribution of drinking water. In Germany we claim that drinking water reservoirs should be in an oligotrophic to mesotrophic state to guarantee a definite supply of drinking water. Despite extensive water treatment there is no certainty of a safe supply of drinking water from an eutrophic reservoir. For this reason, the Wahnbach Reservoir Association constructed a phosphorus elimination plant at the point where the River Wahnbach flows into the reservoir to turn the eutrophic impoundment into a mesotrophic to oligotrophic state by drastically decreasing the phosphorus input.

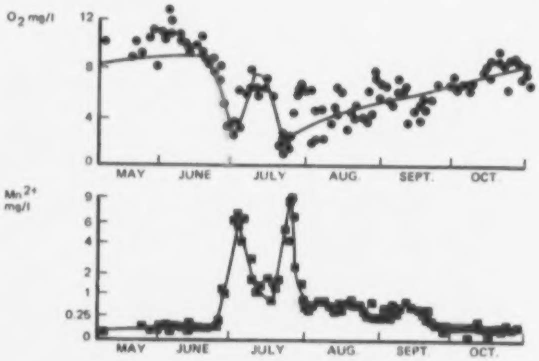

Figure 6. — Connection between the occurrence of Mn^{2+}- ions and the increased O^2-depletion in the water on the bottom of the Wahnbach Reservoir (Investigations, 1969).

ORGANIC CONTAMINANTS IN THE GREAT LAKES

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ABSTRACT

Anthropogenic organic chemicals have been detected in large numbers in the Great Lakes. The major problem has been the accumulation of stable, lipophilic organochlorine chemicals (PCB's, DDT, DDE, mirex) to high concentrations in fish. The presence of these chemicals raises concern over their effects and the recovery time required following implementation of restoration methods. The compounds detected in the Great Lakes and chemical properties related to behavior in the environment are reviewed briefly. A mass balance for PCB's in Lake Michigan is presented. Data are evaluated on the rate of decline of the DDT-group pesticides in coho salmon in Lake Michigan. This data indicate the residence time for DDT in the water column is about 1.74 years. Transport to coho via pelagic and benthic food chains corresponds to approximately 80 to 20 percent, respectively, of the coho body burden of total DDT prior to the ban on DDT use. Transport of PCB's and DDT are expected to be similar. Consequently, reducing or eliminating the input of PCB's and similar chemicals into the Great Lakes should result in a fairly rapid and substantial decrease in the concentrations present in fish.

INTRODUCTION

Contamination by organic chemicals has become a major problem in the Great Lakes (Delfino, 1979). The presence of a wide range of anthropogenic organic chemicals has raised concern over the potential for harmful effects on human health and on the flora and fauna of the Great Lakes ecosystems. The potential adverse effects include reduced reproductive potential, reduced resistance to disease, behavorial abnormalities, cancer, genetic mutations, and physical deformities. Accumulation of some compounds by aquatic organisms to levels considered unsafe for human consumption has been the major health-related problem identified. However, information is sparse on the effects of organic contaminants on aquatic organisms in the Great Lakes.

The problem of contamination by organic chemicals is not restricted to the Great Lakes. Almost all aquatic and terrestrial ecosystems are contaminated to some degree. However, the Great Lakes provide an important example for evaluating the problem. This information can be used to assess the possible behavior of organic chemicals in other aquatic ecosystems.

To illustrate the magnitude of the problem, a partial listing of anthropogenic organic compounds detected in the Great Lakes or their tributaries is given in Table 1. Although this is not an all-inclusive listing, it does suggest the scope of the problem of identifying the compounds, locating their sources, and determining their fate and effects in the Great Lakes.

The most widely documented examples in the Great Lakes are the DDT-group pesticides, polychlorinated biphenyls (PCB's), and mirex. The DDT-group pesticides and PCB's are distributed throughout the Great Lakes at relatively high levels, and Lake Ontario is contaminated with mirex. All three groups of compounds resist degradation, and are lipophilic, leading to high concentrations in pisciverous fishes. These compounds, in particular, have raised concern over the potential for similar behavior by other compounds, especially lipophilic organochlorine compounds. Consequently, a major need exists for understanding the environmental processes and chemical properties controlling organic chemicals and for developing a predictive capability for their transport, distribution, and fate in the Great Lakes.

In this paper some of the environmental processes and properties of organic chemicals controlling their transport and fate in the Great Lakes are discussed and transport processes are evaluated through analyzing data on DDT and PCB's in Lake Michigan. Implications for the rate of recovery from contamination are discussed.

ENVIRONMENTAL PROCESSES AND CHEMICAL PROPERTIES

The amount of an organic contaminant in an aquatic ecosystem can be viewed in terms of the balance between input and loss rates. In general, the rate of change of the concentration of a contaminant in the

Table 1. — A partial listing of organic compounds reported in the Great Lakes and their tributaries, including fish samples.

Compound or Group	Location	Reference
Polychlorinated biphenyls	All Great Lakes	Several, e.g., Veith, 1975; Veith, et al. 1977 Glooshenko, et al., 1976; Swain, 1978 Eisenreich, et al. 1979;
DDT-group pesticides	All Great Lakes	Several, e.g., Reinert, 1970; Veith, 1975 Veith, et al. 1977; Glooshenko et al., 1976; Swain, 1978
Dieldrin	All Great Lakes	Ibid.
Chlordane	Green Bay (L. Michigan) Ashtabula River (L. Erie)	Veith, et al. 1979
Mirex	L. Ontario	Kaiser, 1974, 1978; Holdrinet, et al. 1978
Hexachlorobenzene	Saginaw River (L. Ontario) Grand River (L. Michigan) Tittabawasee River (L. Huron) Maumee River (L. Erie)	Veith, et al., 1979 Ibid.
Chlorobenzenes	Saginaw River (L. Ontario) L. Superior	Ibid. Swain, 1978
Chlorophenols	Fox River (L. Michigan)	Peterman, et al. 1980
Pentachloroanisole	Detroit River (L. Erie) Fox River (L. Michigan)	Veith, et al., 1979 Peterman, et al. 1980
Chlorostyrenes	Ashtabula River (L. Erie) Saginaw River (L. Huron)	Veith, et al. 1979c Ibid.
Chlorobutadienes	Ashtabula River (L. Erie)	Ibid.
Nonachlor	Grand River (L. Michigan)	Ibid.
Polychlorinated dibenzodioxins	Tittabawasse River (L. Huron)	Dougherty, et al. 1979
Polychlorinated dibenzofurans	L. Michigan	Dougherty, et al. 1979
Polychlorinated naphthalenes	L. Michigan	Ibid.
Dehydroabetic acid	Fox River (L. Michigan) Nipigon Bay (L. Superior)	Peterman, et al. 1980 Kaiser, 1977
Polyaromatic hydrocarbons	Fox River (L. Michigan)	Peterman, et al. 1980
Hydrocarbons (petroleum)	L. Michigan	Haile, 1977

waters of the Great Lakes is controlled by the following factors:

1. Inputs
    - (a) Discharges from industrial areas.
    - (b) Municipal waste waters.
    - (c) Land drainage by rivers and streams.
    - (d) Atmospheric wet and dry deposition.
2. Losses
    - (a) Surface water discharge.
    - (b) Volatilization.
    - (c) Sedimentation.
    - (d) Degradation.
    - (e) Harvesting.
3. Recycling
    - (a) Resuspension and/or release from bottom sediment.
    - (b) Biological recycling.

The variety and complexity of the sources and transport routes have been major problems in controlling contamination of the Great Lakes by organic chemicals. Pesticides applied to land areas are transported by air and water. Other compounds produced or used in industrial processes near the Great Lakes or their tributaries, may be transported directly by air or water or stored in various forms, such as chemical stocks, industrial products (e.g., electrical transformers and capacitors containing PCB's), waste disposal sites (e.g., landfills), or in bottom sediments. Storage may be temporary or permanent. Leakage of transformers discarded in landfills may lead to PCB transport by air and water. Contaminants in harbor sediments may be transported gradually by sediment resuspension. However, storage may act as an output for the system. For example, sedimentation in the Great Lakes can bury organic chemicals in the bottom sediments and inhibit their return to the biochemical cycle.

The relative importance of these factors depends on the nature of the organic compound and the lake's characteristics. For specific compounds, the rates of losses by volatilization, sedimentation, and degradation may vary to some extent among lakes due to variations in mass sedimentation rates, concentrations of suspended material, and temperatures of waters and sediments. Furthermore, recycling by resuspension and biological transport may reflect differences in lake morphometry and aquatic organism populations. However, differences in chemical properties are probably more important than lake differences in controlling the rate of change in organic contaminant levels.

Concern over the environmental fate and behavior of organic chemicals has focused attention on the use of physical and chemical properties for predicting environmental behavior. The association of lipophilic or non-polar character with a tendency for persistence and bioaccumulation is widely recognized. Important examples are PCB's and organochlorine pesticides in the Great Lakes. The octanol-water partition coefficient (K_{ow})is often used as a measure of polarity. For many compounds, K_{ow} values are tabulated (Leo, et al. 1971), and K_{ow} values can be obtained fairly readily by direct measurement (Karickhoff, et al. 1979), prediction based on water solubility (Chiou, et al. 1977) or high pressure liquid and chromatography (HPLC) measurements or calculations based on molecular structure (Hansch and Leo, 1979) (Veith, Austin, and Morris, 1979). In turn, K_{ow} values have been correlated with bioconcentration factors (BCF) measured in the laboratory (Neely, et al. 1974; Chiou, et al. 1977; Veith, DeFoe, and Bergstedt, 1979) and with sediment-water partition coefficients (K) for organic compounds (Karickhoff, et al. 1979; Chiou, et al. 1979; Hassett, et al. 1980). Acute toxicity to fish is also highly correlated with K_{ow} . Consequently, a physical property (K_{ow}) shows considerable promise in predicting important aspects of environmental behavior. In the case of adsorption of some compounds by sediments, the K_p of a given compound depends mainly on the organic carbon (OC) content of the sediment. This allows use of an OC-based partition coefficient (K_{oc}) obtained by dividing K by the fractional OC content of the sediment and predicting adsorption from the sediment OC content and the K_{ow} for the organic compound. However, as K_{ow} and OC values decrease (more polar compounds) adsorption estimated from these two parameters may become less accurate.

While obviously important, adsorption and bioconcentration are only two of several processes and factors controlling the behavior of organic chemicals in the environment. Measurements or predictions of hydrolysis, photolysis, volatilization, and biodegradation rates and bioaccumulation through food chain transport are also required. Bioaccumulation from consumption of contaminated food may not be predicted by laboratory measurements of bioconcentration directly from water. Measurements and evaluations of these processes are being actively researched.

Information rates of the important processes controlling environmental distribution and fate can provide the basis for modeling the behavior of an organic chemical discharged into an aquatic ecosystem (e.g., Smith, et al. 1977). While holding promise for assessing the wide range of compounds of environmental concern, such models are at a fairly early stage of development and testing. An alternative approach is the analysis of data on contaminants widely distributed in the environment such as PCB's and DDT.

TRANSPORT OF ORGANIC CONTAMINANTS IN LAKE MICHIGAN

Two basically different approaches can be used to obtain information on the transport of organic contaminants. The first approach is extrinsic in nature, involving the mass balance of PCB's in Lake Michigan. The second, intrinsic, approach involves extrapolating the behavior of PCB's in Lake Michigan from measurements made on certain components of the lake. Both approaches are presented and compared to gain insight into the factors controlling the response of the system to external changes.

A Tentative Mass Balance of PCB's in Lake Michigan

Estimates of PCB loading (input) and losses for a lake can be combined with an estimate of the contaminant "standing crop" to provide some understanding of transport and distribution within the lake (Eisenreich, et al. 1979; Pavlou and Dexter, 1979). Important sources and sinks may be distinguished and contaminant residence time estimated.

The amounts of PCB's stored in Lake Michigan can be estimated based on measurements of the PCB concentrations in the major reservoirs, the lake water, and the bottom surficial sediments. Estimates are summarized in Table 2. Relatively little data are available on PCB concentrations in Lake Michigan

Table 2. — Tentative mass balance of PCB's in Lake Michigan

PCBs in Lake Water (kg)	
Dissolved	8200
Particulate (<20% of total)	1600
Total (2 ng/l)	9800 kg
PCBs in Bottom Sediments (kg)	
0-2 cm layer (0.1 μg/g)	30.000
2-5 cm layer (0.025 μg/g)	30.000
Total	60.000 kg
PCB Inputs (kg/year)	
Atmospheric	
Particulate	1200
Vapor	0 to 2700
Wet	1100 to 4800
Total Atmospheric Estimate[1]	5000
Tributaries (0.05 μg/l)	1650
Industrial Discharges	?
Total	6650 kg/year
PCB Losses (kg/year)	
Volatilization	0 to 3100
Surface water discharge	100
Sedimentation	2600
Total Loss Estimate[2]	2700 kg/year

[1] Assumes wet deposition is 1100 kg/year

[2] Assumes volatilization loss is 0

waters. Murphy and Rzeszutko (1977) found concentrations in the range of 30 to 40 ng/l for a few samples, similar to the concentrations reported by Haile (1977). However, recent unpublished data indicate concentrations may be as low as 1 to 2 ng/l. Although the concentrations are low in either case, the lake water represents a major reservoir of PCB's, and accurate information on the concentrations present is highly important in evaluating fluxes, residence times, and the rate of response of the system to changes in external loadings. Based on a concentration of 2 ng/l and a lake volume of 4.9×10^3 km^3 (Klein, 1975), the estimated mass in the lake water is about 9,800 kilograms. The sediment-water partition coefficient for PCB's is estimated to be about 1×10^5 (Karickhoff, et al. 1979; Pavlou and Dexter, 1979). Based on this partition coefficient and suspended particulate matter concentrations in the range of 0.5 to 2 ng/L (our recent measurements), less than 20 percent (1,600 kg) is expected to be present as "particulate" PCB's.

The bottom sediments also represent a major reservoir of PCB's. Our recent data indicate PCB concentrations in surficial sediments in southern Lake Michigan range from 0.005 to 0.2 μg/g. These are the same range as concentrations reported in Lake Superior sediments (Eisenreich, et al. 1979). The variations in concentration with location in southern Lake Michigan are apparently related in part to variations in sedimentation rate and depth of mixing of the surficial sediments. Based on estimated average PCB concentrations (dry-weight basis) of 0.1 μg/g and 0.025 μg/g and sediment porosities of 75 and 60 percent for the 0 to 2 and 2 to 5 cm layers, respectively, the estimated sediment PCB reservoirs are 30,000 kg in the 0 to 2 cm layer and 30,000 kg in the 2 to 5 cm layer.

The major sources of PCB's to the lake are atmospheric deposition, tributaries, and industrial discharges. Recent information indicates atmospheric input is important and may be the major source of PCB's to Lake Michigan. Based on field measurements, Murphy and Rzeszutko (1977) estimated the input in precipitation was 4,800 kg/year. Atmospheric input by both wet and dry deposition was estimated by Doskey (1978). Wet deposition was estimated to be about 1,100 kg/year based on measurements of PCB concentrations in air over Lake Michigan and calculated PCB washout. This lower value is used to calculate the PCB input in Table 2. The estimated atmospheric input of particulate PCB's was 1,200 μg/g based on measured concentrations and an estimated deposition velocity. Uncertainty exists over the vapor input because of uncertainty in the Henry's Law constant (air-water partition coefficient) for PCB's and, thus, whether air-water transfer is gas-phase or liquid-phase controlled (Doskey, 1978). However, laboratory measurements indicate transfer is probably gas-phase controlled. This means net vapor transfer would be from air to water; the estimated input is 2,700 kg/year. Consequently, the combined atmospheric input is about 6,650 kg/yr. The wide distribution of PCB's in the environment is consistent with the importance of atmospheric transport. Examples are present of PCB's in Lake Superior (Glooshenko, et al. 1976; Veith, et al. 1977; Swain, 1978; Eisenreich, et al. 1979) and the north Atlantic (Harvey and Steinhauer, 1976).

Tributary inputs of PCB's to Lake Michigan are probably less than the amounts received from the atmosphere. However, comprehensive data on tributary inputs are lacking, partly because of the large number of tributaries and expected variations with both location and time. In 1970 to 1971, PCB concentrations ranging up to 0.45 μg/l were observed in tributaries entering Green Bay (Veith, 1972). Municipal and industrial wastes are known to be sources of PCB's in tributaries. For example, PCB's were detected in effluents from wastewater treatment plants in the Milwaukee River watershed (Veith and Lee, 1971), southeastern Wisconsin (Dube, et al. 1974), and Michigan (Hesse, 1976). PCB's were also found in effluents from pulp and paper mills (Kleinert, 1976; Peterman, et al. 1980). Inputs to tributaries from these sources may be declining with decreasing use of PCB's. However, leaching from landfills and other discharges associated with disposal may represent continuing sources. Based on the available data on concentrations in streams and wastewater effluents, Murphy and Rzeszutko (1977) estimated the input to Lake Michigan from these sources was approximately 1,650 kg/year. For an average tributary flow of 33 km^3/year, this would correspond to an average PCB concentration of about 0.05 μg/l .

Losses of PCB's from the lake system occur through surface water discharge and permanent sedimentation. Biodegradation, volatilization, and harvesting losses are considered negligible. Although PCB's can be partially degraded by microorganisms (Furakawa and Matsumura, 1976), our laboratory experiments (Flotard, 1978) involving incubation of Aroclors in sediments and measuring changes with time in major peaks indicated negligible degradation in sediments. More recent experiments in our laboratory showed some degradation of low-chlorine PCB's in sediments but indicated degradation was retarded by PCB adsorption on sediments. In Lake Michigan sediments, adsorption and low temperatures may completely inhibit degradation. Volatilization is also uncertain. If air-water transfer were liquid-phase controlled, volatilization could amount to 3,100 kg/year in Lake Michigan. However, the evidence for gas-phase control indicates volatilization losses may be negligible (Doskey, 1978). Losses through harvesting are slight because of the small proportion of PCB's contained in the fish population. The loss through surface water discharge (water residence time 100 years) is about 98 kg/year based on the water concentration of 2 ng/l and an outflow rate of 49 km^3/year.

The major mechanism for PCB removal from the system is sedimentation and burial in the bottom sediments. Transport of PCB's to the bottom sediments can be estimated from the mass sedimentation rate and the concentration of PCB's in the depositing sediment. The average mass sedimentation rate for the southern basin of Lake Michigan has been estimated to be 7 mg/cm^2/year based on ^{210}Pb measurements (Edgington and Robbins, 1976). Allowing for decomposition of organic matter after deposition (assumed to be 50 percent), a mass deposition rate of about 15 mg/cm^2/year is obtained. Our recent data indicated the PCB concentration in surficial fine-textured sediment in Lake Michigan is about 0.2 µg/g. Allowing for some dilution by mixing with non-contaminated sediment, an estimated PCB concentration in depositing sediment of 0.3 µg/g is obtained. This is probably high. For example, calculations based on a sediment-water partition coefficient of 1 × 10^5, a suspended sediment concentration of 1 mg/l, and a lake water total PCB concentration of 2 ng/l indicates the concentration in the suspended (depositing) sediment should be about 0.18 µg/g. However, if the mass sedimentation rate for the southern basin is assumed to represent the average for the entire lake, the calculated PCB deposition rate based on a mass deposition rate of 15 mg/cm^2/year and a PCB concentration of 0.3 µg/g is about 2,600 kg/year.

The mass balance calculations indicate either the system is not in steady-state with respect to PCB's or the mass balance is in error. The estimated input to the water column (6,650 kg/year) exceeds the loss (2,700 kg/year). Furthermore, if PCB input is assumed to have occurred at this rate over a 20-year period, the estimated inputs (~133,000 kg) exceed the estimated amounts in the water and sediments (69,800 kg) plus the amounts lost by surface water discharge (100 kg) or about 70,000 kg. Several possible explanations exist for these discrepancies:

1. The inputs may be high.
2. The lake water and/or sediment concentrations of PCB's may be low.
3. The sedimentation rate for PCB's may be low.
4. Biodegradation and/or volatilization may contribute to PCB losses.

As discussed previously, some uncertainty exists over the importance of biodegradation and volatilization losses. However, available evidence supports the assumption of negligible losses by these pathways. While the calculated sedimentation rate for PCB's could be low evidence suggests the value is probably high. Estimates for lake water and sediment PCB concentrations are conservative, therefore, the estimates of PCB's stored in these in-lake reservoirs may be low. Consequently, the most likely explanations for the discrepancies seem to be an underestimate of lake water and sediment values and/or an overestimate of input rates. The loading rates do not include any estimates for contributions from point sources such as the previous industrial discharge at Waukegan (Murphy and Rzeszutko, 1977). Consequently, inputs from other sources may be overestimated.

If the systems were in approximate steady-state, the apparent residence time for PCB's in the lake water could be calculated from the amount of PCB's (9,800 kg) in the lake water and the PCB loss rate (2,700 kg/year), or the input rate (6,650 kg/year). The residence time would be 3.6 years based on the loss rate and 1.5 years based on the input rate. More accurate estimates of PCB input, losses, and storage are needed to estimate residence times using the mass balance approach.

Analysis of Data on t-DDT and PCB Concentrations in Fish

Fish accumulate microcontaminants directly from water via their gills (direct uptake) and from their food (consumptive uptake). Direct uptake can be responsible for efficient bioconcentration of compounds which are not eliminated by fish. Long-term laboratory studies have shown brook trout can accumulate PCB's to levels 8,000 to 25,000 times ambient water concentrations (Snarski and Puglisi, 1976). Such findings agree with the 2,4,5,2′,5′-PCB bioconcentration factor (BCF) of 14,500 expected for rainbow trout based on aqueous solubility-BCF correlations presented by Chiou, et al. (1977).

Although direct bioconcentrations of these magnitudes are dramatic, they do not account for the fish PCB levels found in the environment. If PCB's were irreversibly accumulated by Lake Michigan lake trout solely from a concentration of about 2 ng/l "dissolved" in water, the expected concentrations of PCB's in these fish would be between 0.016 and 0.05 ppm, based on these BCF's. A bioenergetics-based model relating direct PCB exposure to oxygen uptake demonstrated that the amount of PCB reaching the gills of an adult Lake Michigan lake trout at a lake water PCB concentration of 1 ng/l could account for a maximum whole fish concentration of 0.09 ppm (Weininger, 1978). Lake trout in Lake Michigan contain PCB levels of 15 to 35 ppm (Willford, 1977; Veith, 1975; Weininger, 1978).

This implies that lake trout receive PCB's primarily from their foods. A simple approach to evaluate the importance of food chain biomagnification of microcontaminants involves comparing the amount of contaminant ingested to the achieved growth of an organism. Dividing the contaminant concentration in the diet by the gross conversion efficiency (GCE) of growth for the fish provides a measure of the maximum expected contaminant level in the fish attributable to dietary accumulation. Adult lake trout in Lake Michigan feed primarily on adult alewives containing 4 to 7 ppm of PCB (Eck, 1977; Veith, 1975; Weininger, 1978). The mean gross conversion efficiency of these trout has been estimated to range between 23 percent (2 to 3 years old) and 14 percent (7 years old) (Weininger, 1978; Stewart, et al. 1980). Assuming a diet containing 5.5 ppm of PCB, lake trout are expected to accumulate as much as 24 to 40 ppm of PCB via dietary exposure. The data showing PCB concentrations in lake trout are in this range support the conclusion that dietary accumulation is predominantly important in this system.

Recognizing PCB and PCB-like contaminants are principally transported to predatory fish via the food chain, two primary pathways can be described (see Figure 1). The first is a pelagic pathway: water → phytoplankton and suspended particulates → zooplankton → macroinvertebrates → forage fish → pisciverous fish. The portion of contaminants currently reaching fish via the pelagic pathway has never been removed from the water to the bottom sediment; this portion is expected to have a fairly short residence time in the water column. A second, benthic pathway may also exist: water particulate matter → sediment → benthic invertebrate → forage fish → pisciverous fish. The portion of a contaminant transported in this manner comes from a sediment "reservoir". In lakes where the sedimentation rate is low, the benthic pathway is expected to continue to make persistent contaminants available to the lake biota for a long time.

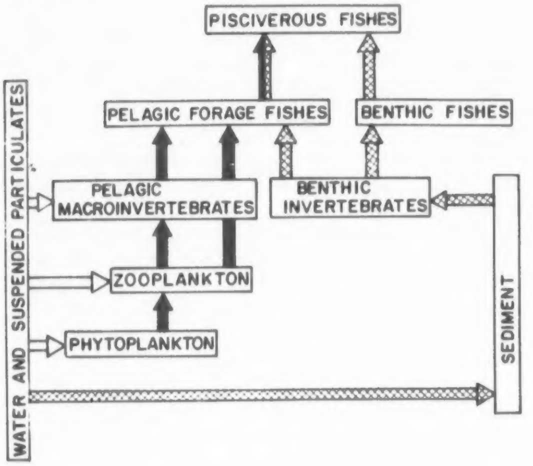

Figure 1. — Pelagic (solid) and benthic (cross-hatched) pathways of contaminant transport to pisciverous fishes.

We evaluated data on the t-DDT levels in Lake Michigan coho salmon to obtain insight into the relative importance of the pelagic and benthic pathways. The conclusions from this analysis have been summarized briefly elsewhere (Francis, et al. 1979). A more detailed analysis follows. The widespread use of DDT resulted in a high degree of contamination of Lake Michigan fishes during the 1960's. In 1970, the use of DDT was banned. DDT degrades to a limited number of products; by examining the total concentration of DDT and its degradation products (t-DDT) during subsequent years, information on the transport of such chemicals in the environment can be obtained. Coho salmon are short-lived, fast growing fish and feed almost exclusively on alewives during their adult years. Contaminant concentrations in coho salmon are therefore expected to respond rapidly to changes in the levels of environmental contamination.

Following the DDT ban in 1970, t-DDT concentrations in coho decreased rapidly. However, the t-DDT concentrations in coho seem to approach a new level distinctly higher than zero (Figure 2). It is hypothesized that these concentrations result from a two-part phenomenon. The rapid decrease reflects the removal of t-DDT from the lake water column and corresponds to the direct and pelagic food chain transport of t-DDT to coho. Under these assumptions, "pelagic" portion of t-DDT transport can be modeled simply as an exponential decrease with time. The second part of the model reflects the benthic transport route. Since sedimentation in Lake Michigan is low and the age of the sediment mixed zone is high (Robbins and Edgington, 1975), a very slow decrease in transport via this route is expected. Although transport from the sediments will decline slowly due to gradual burial of PCB's in the sediments, the benthic pathway contribution is modeled here as remaining constant. The simplified model is thus written:

where y = t-DDT concentration in coho salmon
t = time, in years since DDT input elimination
a, b, c = constants.

A weighted non-linear regression provides the following equation of best fit:

$$y = 11.8 \exp(-.575t) + 2.59$$

Figure 2 shows this model as well as the individual components. The portion of the 1970 t-DDT levels in coho salmon attributed to the benthic pathway (c) is about 18 percent. The residence time of t-DDT in the water column (1/b) appears to be short (1.74). This is attributed to t-DDT removal by sedimentation, i.e., adsorption of t-DDT by the depositing particulate matter.

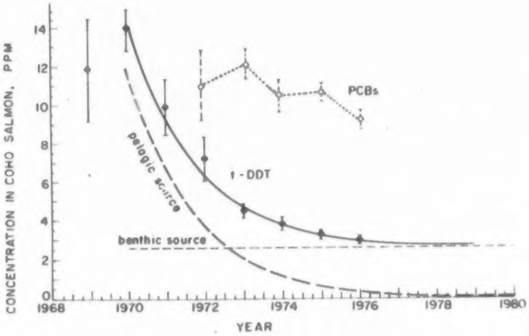

Figure 2. — Concentrations of t-DDT (◆) and PCBs (◆) in Lake Michigan coho salmon and 95 percent errors (vertical lines). Data from Willford (1977). Solid line shows result of weighed nonlinear regression model; dashed lines show estimated pelagic and benthic components.

Comparing Results

The short pelagic residence time obtained for t-DDT in Lake Michigan (1.74 years) agrees approximately with the residence times (1.5 to 3.6 years) for PCB's estimated by the mass balance approach. PCB's and t-DDT are expected to behave similarly with respect to sedimentary removal. This supports the validity of assumptions made in calculating the mass balance. The 1.75-year residence time calculated from observed coho levels is thought to be accurate; this calculation procedure would tend to result in over- not under-estimation of residence time. Reports of similar residence times for other insoluble substances (Koide and Goldberg, 1961) further support this contention. Unfortunately, time series data on the t-DDT level in Lake Michigan water, which would verify this result, are not available.

Differences in the behavior of t-DDT and PCB's could lead to some differences in their residence times in Lake Michigan water. Differences in suspended particulate matter-water partition coefficients might lead to differences in sedimentation rates and residence times. However, the partition coefficients are in the same range (Chiou, et al. 1979). Differences in volatilization rates and/or degradation rates are also possible. In aerobic systems, DDT degrades to DDE (see Guenzi and Beard, 1976a) but DDE is stable. In anaerobic sediments, DDT degrades through DDD to other products not measured as t-DDT (Guenzi and Beard, 1976b). Considering the aerobic nature and low temperature of Lake Michigan bottom waters, the degradation of DDT to DDD in Lake Michigan sediments is probably slow. If t-DDT degradation in sediments between 1970 and 1976 was significant, part of the decline in coho t-DDT levels could have resulted from decreasing sediment t-DDT levels and decreasing transport via the benthic pathway. This would result in an underestimate of the t-DDT residence time in the lake water, i.e., some of the decline in coho t-DDT levels would be caused by declining t-DDT levels in the surficial sediments. However, this effect would be small because the decline in lake water levels is relatively rapid and the proportion of the 1970 levels accumulated from the water column is relatively large.

The PCB mass balance is tentative because of uncertainties in input and output magnitudes. If the system is approximately in steady-state and the residence time for PCB's in the Lake Michigan water column is 1.74 years, the PCB loading or loss required to maintain the observed amount of PCB's in the water column can be calculated:

$$\text{annual loading or loss} = \frac{\text{standing crop}}{\text{residence time}} = \frac{9{,}800\ \text{kg}}{1.74\ \text{years}} = 5{,}632\ \text{kg/year}$$

This value falls between the estimated loading (6,650 kg/year) and loss (2,700 kg/year) calculated in the mass balance (Table 2). This loading is based on the lower range of reported estimates of atmospheric input by wet deposition (1,100 kg/year). The general agreement between the input-output rates based on the t-DDT residence time and mass balance approaches supports the validity of the calculated mass balance while illustrating the lack of precision involved in its calculation.

The PCB levels in Lake Michigan coho salmon have not shown the decline observed for the t-DDT levels (Figure 2). Similar observations have been reported for Cayuga Lake, N.Y. (Wszolek, et al. 1979). Assuming similar transport for the two groups of compounds, this indicates the input of PCB's to the Lake Michigan system has not been dramatically reduced, although the use of PCB's and DDT was limited at nearly the same time. (DDT was banned from use in 1970; use of PCB began to decline in 1971-1972). This indicates available reservoirs of PCB's exist in the harbors, rivers, drainage ditches, and landfills in the Great Lakes Basin. Apparently, transport of PCB's from these reservoirs to Lake Michigan is continuing.

MANAGING THE USE OF NEW CHEMICALS

Approximately 1,500 new chemicals per year are created and marketed for a wide variety of industrial, agricultural, and domestic uses. Measures are needed to ensure that they do not become future contaminants. An adequate system of screening chemicals must be developed by Federal agencies in the near future. To be effective, the screening process should be mated to internationally consistent certification procedures. The rewards for efforts in this regard can be expected to be indirect, but substantial; it is ultimately cheaper to refrain from polluting than to restore an ecosystem.

A variety of methods might be used to evaluate the hazards of new chemicals (Dickson, 1979). OECD in Europe and the Office of Toxic Substances in the United States are developing a tier-structured screening program similar to that proposed by Cairns (1980). Depending on the proposed use, a chemical would be required to pass a set of tests. The first-tier (screening) tests are rapid and relatively inexpensive. Proposed screening tests include the BOD test (a chemical should show at least 60 percent of its theoretical biochemical oxygen demand), acute toxicity tests (LC for *Daphnia* and fathead minnows), the Ames test, and a test for photosynthetic inhibition activity. If a chemical does not pass a screening test, or if its proposed usage warrants, further more sensitive (and more expensive) tests may be required. These might include a long-term rodent test for carcinogenicity, chronic toxicity tests with fish (embryo-larval growth), and a test for bioaccumulation and persistence potential. The most extensive tests are reserved for chemistry which will be introduced into the environment such as high usage industrial chemicals. These tests are expected to be primarily field studies and will be quite expensive.

The responsibility for conducting certification tests for new chemicals falls to industry. In most cases, chemical industries have tried to develop and use safe substances or to recommend adequate disposal techniques to their industrial customers. Their mandatory participation in certification screening is a logical extension of this effort.

Control of substances currently in use (and for which disposal permits are already issued) is the responsibility of Federal, State, and Provincial governments. The success of such a program will depend upon the establishment of consistent usage-related certification and availability of adequate disposal sites for hazardous wastes. Chemicals in current use can be screened by the same methods proposed for new chemicals; consideration should be given both to uses and to disposal. Disposal sites must be made available and enforcement procedures established to ensure their use. Disposal site adequacy can be insured only by appropriate monitoring.

MANAGEMENT TECHNIQUES FOR CONTROLLING PERSISTENT POLLUTING SUBSTANCES IN THE GREAT LAKES

The approaches available for reducing the levels of contaminants in the Great Lakes are limited mainly to various forms of loading reductions. Reducing the contaminant loading to a lake will be followed by a rapid reduction in the contaminant levels in the lake's fish populations (e.g., DDT in Lake Michigan coho salmon). The long-term residual contamination of a system appears to result chiefly from benthic recycling. Even in the case of t-DDT, the benthic contribution is small (<20 percent of the 1970 levels).

The short residence time of insoluble organic compounds demonstrates the potential for large lakes to rapidly clear their pelagic zones of these compounds and justifies a comprehensive program to remove contaminant sources. The first part of this program must consist of a systematic effort to identify the point sources of pollutants on a harbor-by-harbor, river-by-river basis. Removing these sources by both discharge elimination and contaminant-reservoir containment (dredging) or destruction (incineration) can provide substantial rewards in a short time.

Diffuse sources, such as landfills, are difficult to control, and are expected to continue to release volatile contaminants, such as PCB's, for many years. The potential for reducing atmospheric sources is high because their residence time in the atmosphere is relatively short (20 to 60 days; Bidelman and Olney, 1974). However, lateral atmospheric transport is rapid, and truly effective control requires worldwide compliance. Current stores of organic contaminants must be identified and practical methods of disposal developed.

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ORGANOCHLORINATED COMPOUNDS IN DRINKING WATER AS A RESULT OF EUTROPHICATION

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ABSTRACT

The impact of eutrophication on drinking water supplies may be serious. It makes its preparation more difficult and costly. Moreover, it generally reduces the final quality of drinking water distributed, by giving rise to unpleasant and persistent taste, as well as leading to the formation of hazardous organochlorinated compounds. Eutrophied waters are, in general, very rich in organic substances, arising from the metabolism and decay of algae and other aquatic plants. When any chlorination treatment is applied, organic substances react readily with chlorine, forming soluble organochlorinated compounds which are very persistent and cannot be efficiently removed. The high content of organic and nutrient substances contributes to the "dirtiness" of the distribution network and may increase risk of bacterial growth which also encourages higher final chlorine application. The formation of organochlorinated compounds will continue in the distribution system as long as both chlorine and organic substances persist. Although organochlorinated compounds may already be present as pollutants in raw waters,the chlorination treatment itself is usually by far the main source of these substances in drinking water. Trihalomethanes are the volatile compounds, and can easily be identified, but on average they represent only a modest proportion (20 percent) of all organochlorines in drinking waters. The non-volatile compounds (up to 80 percent of organochlorines) are still very poorly identified but may well contain more hazardous compounds. The health risk (essentially cancer) from organochlorines, cannot yet be fully evaluated but toxicity tests and epidemiological studies suggest that extensive measures need to be urgently considered to prevent their presence in drinking waters.

INTRODUCTION

Eutrophied waters contain a substantial quantity and variety of organic substances arising mainly from the metabolism and decay of algae and other aquatic plants. These substances may cause unpleasant taste but in general are not directly toxic to man. Nevertheless, the utilization of eutrophied raw waters generally substantially increases the chlorine dosage and use in drinking water treatment. Instead of a moderate application for disinfection, at the end of treatment, chlorine may be used extensively throughout the whole system: (a) during raw water transportation to prevent the growth of fixed organisms in the pipes; (b) during the treatment itself to control organisms, breakdown of ammonia and other substances, etc.; (c) as a final disinfection; and (d) to maintain an increased chlorine residual in the distribution system (because of the increased consumption of chlorine by the organic substances still present). Eutrophied waters, because of their high organic content and increased chlorine treatment, clearly encourage increased formation of organochlorinated compounds.

EXTENT OF THE PROBLEM

The primary concern in traditional drinking water treatment has been to control micro-organisms which cause waterborne diseases (such as typhoid and cholera) and to provide an aesthetically acceptable water (taste, odor, color). This goal has been achieved largely by using chlorine and other oxidants in conjunction with other treatment processes. Recently, however, the presence of chemical pollutants in drinking water and their possible health hazards have caused increasing concern. With new analytical techniques and instrumentation, such as gas chromatography and mass spectrometry, several hundred specific organic pollutants have been identified in low concentrations* in various drinking water supplies. These compounds generally originate to a minor extent from the polluted raw waters but to a larger extent from the drinking water treatment itself. Concentrations of these pollutants vary from virtually nil in drinking water drawn from protected ground water to substantial amounts in drinking water derived from contaminated surface and ground waters which are chlorinated.

Potable water treatment may considerably increase the content of synthetic chemicals in drinking water; recent studies in many countries indicate that the large number of halogenated products formed by chlorination are often a major portion of the indentifiable synthetic chemicals in drinking water. These by-products are found especially in drinking water derived from water containing precursors (such as eutrophied waters) when treated with chlorine; they can be present at concentrations of up to several hundred micrograms per litre. The formation of organochlorines, in waters rich in organic precursors, is roughly proportional to the extent and intensity of the chlorine treatment. For instance, compared to the same treated waters receiving no chlorine application, a chlorine disinfection alone may increase the organochlorinated compounds by 10 times or more, and breakpoint chlorinations 100 times or more (see Table 1).

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* Typically from 0.01 to 100 microgram/litre (ug/l).

Table 1. — Effect of chlorination on the occurrence of some halogenated compounds in tapwater, concentration range in μg/l (data from the Netherlands).

Parameters	Type of treatment with chlorine		
	None	Final disinfection only	Breakpoint chlorination
Number of supplies studied	13	4	3
Chloroform	0.01 - 2.0	0.1 - 10	25 - 60
Bromodichloromethane	0.01 - 0.9	0.1 - 10	15 - 55
Dibromochloromethane	0.01 - 0.1	0.01 - 5	3 - 10
Dichloroiodomethane	0.01	0.01 - 0.3	0.01 - 10
Bromochloroiodomethane	0.01	0.01 - 0.03	0.01 - 0.3
Bromoform	0.01	0.01 - 1.0	3.0 - 10
1, 1 Dichloroacetone	0.005	0.005	0.1 1.0
Trichloronitromethane	0.01	0.01 - 3.0	0.01 - 3.0

The organohalogens (see Appendix) formed are mainly organochlorinated compounds but brominated and iodinated compounds can also be present. Only a portion (about 20 percent) of the organohalogens present in drinking water can currently be identified (Figure A). These are mainly the volatiles such as the trihalomethanes (THM's) which include chloroform. The other identified compounds which may originate from the raw water account for about 2 percent but represent a large number of compounds (chlorophenols, PCB's, pesticides, etc.). The non-volatile compounds (up to about 80 percent) are difficult to identify with current analytical techniques (gas chromatography, mass spectroscopy). They represent a large number of compounds and some may be of greater toxicological significance than the identified portion (THM's). Their overall level in water can be measured by the TOCl test (Total Organic Chlorine). The total amount of organohalogens reaching the consumer may be higher than the amount measured in the water leaving the treatment plant, because these chemicals continue to form in the distribution system as long as precursors and chlorine are present. Byproducts may also be formed when using alternative oxidants such as ozone or chlorine dioxide but probably to a lesser extent; very limited knowledge exists on these.

Knowledge of the relationship between the trophic state of waters, the production of organic precursors, and the potential formation of organohalogens still seems to be relatively modest. Basic factors determining the "yield" of organohalogens during drinking water treatment are not only the quantities of chlorine and organic precursors, but also the pH and temperature.

The following figure is intended to illustrate a typical distribution of organohalogenated compounds that may be found in raw waters drawn from rivers in industrialized countries and in treated waters (after chlorine treatment).

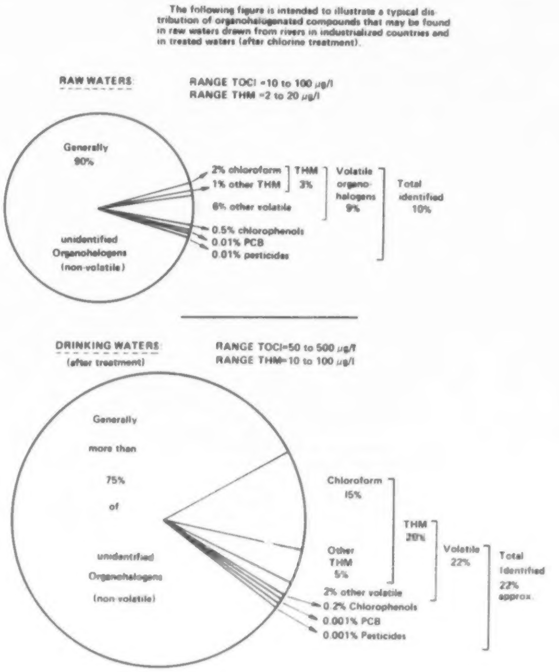
The following figure is intended to illustrate a typical distribution of organohalogenated compounds that may be found in raw waters drawn from rivers in industrialized countries and in treated waters (after chlorine treatment).

Figure A. — Typical distribution of organohalogens in waters in industrialized countries.

1. *Chlorine*: As already stated, an increase in the trophic level of the raw waters used generally increases chlorination application: in transportation of raw water, and in treatment, disinfection and distribution of drinking water.

2. *Precursors*: Eutrophied water is, of course, much richer in a variety of organic precursors. Humic substances (decay of cellulose and lignin) are generally the main ones; chlorophyll and its derivatives are also precursors.

3. *pH*: Under eutrophic conditions, algal activity tends to consume the CO_2 present in water which alters the carbonic equilibrium, with a corresponding rise in the pH (which can reach 9). Moreover, a higher pH increases the yield of volatile organohalogens (trihalomethanes). Under certain conditions, with the rise of 1 unit of pH, this yield may double. Little is known so far about the corresponding variation of the non-volatile organohalogens. It seems that the proportion of volatiles in drinking water (approx. 25 percent) vis-a-vis the non-volatiles (75 percent) tends to increase with a pH rise. However, the question is: do the non-volatiles really decrease in absolute value, remain stable or even slightly increase with a pH rise? More knowledge on this point would be desirable.

4. *Temperature*: A rise in temperature increases the yield of organohalogens (for instance at 20°C the yield of organohalogens is about 50 percent higher than at 4°C).

The temperature factor shows that in summer the conditions for organohalogen formation are at their maximum and strategies to control organochlorine compounds should pay special attention to this fact. In practical terms, the storage of water before treatment has proved to be useful in decreasing the content of various pollutants (including organic precursors). However, the organic precursors may rise again rapidly (after 15 days, for instance) as a result of eutrophication, especially in summer.

The parameters currently used in most countries for controlling drinking water treatment plants do not generally include measuring contamination by organics and organohalogen byproducts. There is a clear need for suitable test procedures and also for operational control of the processes to minimize the formation of these byproducts. So far, four analytical techniques are available to measure actual or potential organic contamination: (1) Total Organic Chlorine, (2) trihalomethane analysis, (3) Total Organic Carbon, (4) oxidant demand measurement.

Techniques 3 and 4 give no indication of the byproducts that may be formed during treatment with oxidants but measure the precursors present in the water. Trihalomethane analysis is becoming widely known and is within the analytical capability of at least major water treatment works' laboratories. However, it measures only a small part of the total halogenated byproducts formed, and could be considered as a "marker." TOCl is the most comprehensive and relevant test and should be developed as a standard test (it does not indicate, of course, which individual chlorinated compounds are present).

At the low individual concentrations at which some organic compounds may occur in drinking water, the primary concern is for their potential contribution to chronic health risks, e.g., cancer. Although the specific causes of cancer are not yet fully understood, there is growing agreement among scientists that exposure to carcinogenic contaminants in man's total environment which include food, water, and air, may contribute to the incidence of cancer which accounts for up to one-third of the annual mortality in OECD countries. Many organohalogenated compounds may be found in drinking water at low concentrations. Even at the concentration of some micrograms per litre, the aggregate exposure to such chemicals from a lifetime of water consumption contributes a potential risk to human health. In addition, not only is the exposure to each of these compounds separately of concern, but also the possibility of synergistic effects. Furthermore, certain sections of the population are at greater risk because of age, physical state, environmental stresses, and possibly genetic disposition.

The assessment of the effects of synthetic organic chemicals on man is mainly based on animal tests and on epidemiological studies using statistical data on human diseases and mortality. In 1976, the U.S. National Cancer Institute published a study which showed that under laboratory conditions, cancer was caused in rats and mice by daily exposure to high doses of chloroform. Long-term toxicity tests carried out in France on mice and rats with organic micropollutant extracts from chlorinated drinking water, showed a significant increase in the incidence of various types of malignant tumors.

Various epidemiological studies have explored the association between organohalogens, or some surrogate parameter found in drinking water, and various types of cancer. Epidemiological investigations in the United States have indicated correlations between increased cancer rates and areas where poor quality, chlorinated surface waters supply the drinking water. An epidemiological study in the Netherlands of 4.6 million inhabitants has suggested that where drinking water is prepared from surface waters of poor quality, which are chlorinated, a higher cancer mortality rate was found (especially esophagus and stomach) than in areas where it is prepared from ground waters of good quality and generally not chlorinated. Although it is not yet possible to fully evaluate and quantify the health hazard resulting from drinking water chlorination, it is thought that there may be no "safe" or "no-effect" levels for organohalogens. Other than estimates on health risks from chloroform, knowledge is still lacking on the potential hazards from the large number of other unidentified organohalogenated compounds in water. Thus prudence is required and it is justifiable to maintain organochlorine concentration as low as feasible in drinking water supplies.

TREATMENT AND DISINFECTION PROCESSES

Evaluation of Possible Approaches

Over the past few decades, potable water supply has generally been characterized by (1) a net decrease in the quality of many raw waters used (pollution, eutrophication), and (2) the consequent intensification of the treatment applied. The parallel increase of organic pollutants in waters and chlorine levels used in treatment (such as breakpoint chlorination) has led to high organohalogen concentrations in a number of drinking water supplies.

Unfortunately, the current practice in many drinking water treatment plants is still to use chlorine extensively throughout the system. Although its use corresponds to specific functions, organohalogen formation will take place all along the system. Any realistic control policy should carefully consider these stages:

1. *Raw water transportation:* Chlorine is used here for its biocidal effect, i.e., to prevent growth of fixed organisms in the mains. Other techniques can be used such as preliminary filtration and clarification of raw waters before transportation, mechanical cleaning, etc.
2. *Purification treatment:* Oxidants are used here for several purposes:
   a. The oxidant effect is aimed mostly at removing various organic and inorganic contaminants such as ammonia and substances causing taste, odor, or color. Breakpoint chlorination is frequently carried out to remove ammonia but various alternatives can be applied such as biological removal, storage in reservoirs, or ion exchange. Color can often be effectively removed by coagulation, and powdered activated carbon dosing usually controls taste and odor.

b. The biocidal effect is used in different parts of the treatment (filters, settling tanks, etc.) to prevent growth of algae and other organisms. High chlorine doses may be used, especially in summer, for this purpose. Various alternatives (physical, mechanical, or chemical) exist for this application.

c. Other miscellaneous effects, such as action on colloids and sludge, can be replaced by other approaches.

3. *Disinfectant effect:* This is the primary purpose for which chlorine and other oxidants are used:

a. For waters drawn from polluted sources, disinfection is necessary. Various approaches exist: The use of an oxidant such as chlorine, ozone, chloride dioxide, etc., or ultra-violet treatment. Various filtration techniques such as slow sand filtration, bankside filtration, surface infiltration (on soil or dunes) are very effective and substantially reduce the need for disinfection;

b. For waters drawn from unpolluted and well-protected sources (this is the case for ground waters especially). Different viewpoints exist, however, in various countries: (1) in a number of countries it is judged that systematic chemical disinfection of waters of good biological quality is unnecessary; thus often it is not applied; (2) in a few countries, however, regular chlorine disinfection is applied systematically, even to high quality waters.

4. *Residual "bacteriostatic" effect in the distribution network:* This is also a controversial point. In some countries, it is not common practice to maintain a chlorine residual in the distribution network under normal conditions, although this may be done on some occasions (when a network is not in a good state of maintenance for instance). Other persistent disinfectants can also be used such as chlorine dioxide or chloramines. A clean and well-maintained network is a desirable policy to minimize application of a final disinfectant.

The different oxidants used as treatment reagents and disinfectants have advantages and disadvantages, both in terms of their effectiveness and the byproducts they may generate. The main alternatives to chlorine, which have already been used in full-scale operation over a certain period and for which experience exists, are ozone and chlorine dioxide. Ozone has been used for potable water disinfection since the beginning of this century. It is an efficient oxidant and a powerful disinfectant but does not leave a residual in the distribution system; therefore, where necessary, a bacteriostatic agent (such as chlorine dioxide, chloramine or chlorine) may be added. Chlorine dioxide is also an efficient oxidant and a very good disinfectant; it leaves, like chlorine, a residual in the distribution system. It does not remove ammonia. Little is known about the possible byproducts of using ozone and chlorine dioxide and there is concern about the chlorite and chlorate generated when using chlorine dioxide. Ozone and chlorine dioxide are more satisfactory for taste and odor problems than chlorine and have been used for this reason.

The cost of water treatment is generally a small fraction of the consumer's cost for drinking water. In many cases minor modifications to existing treatment aimed at minimizing the precursors before applying oxidants and optimizing oxidant application without endangering the biological quality of the water, will be effective for little or no cost in substantially reducing byproducts. As the cost involved is usually moderate, it is prudent to carry out these modifications where feasible. Using certain treatments such as granular activated carbon or resins to remove organochlorine byproducts after formation would be by far the most costly option, and probably only needs to be considered in those cases where water quality is so poor that other conventional technologies cannot sufficiently reduce oxidant demand and precursors. Control options available to small water systems differ considerably from those available to large systems because small systems have higher per capita costs, less access to trained operating personnel, and less capacity to monitor sufficiently. Using high quality raw waters is thus particularly important in this case as it makes the whole treatment and distribution far easier and safer.

Alternative Approaches

Controlling organohalogens in drinking water involves either preventing their formation, or removing them after they have been formed. The latter approach is not at present practicable since organochlorines, once formed, are generally very persistent and pass through conventional treatment. The preventive approach is the safer and better method and in general may be achieved in the following ways:

1. By encouraging the selective use of raw waters of better quality (non-polluted and non-eutrophic). Where this is feasible, the use of chlorine (or other oxidants) can be avoided or at least minimized.

2. By using alternative purification processes (filtration, precipitation, etc.) which minimize the use of chlorine or other oxidants at any stage. This approach is particularly advisable in the case of raw waters which are moderately or not polluted.

3. By minimizing the dose of chlorine applied and limiting its use to final disinfection only. This approach may be practical in many situations (small water supply installations, for instance) and will be easier if combined with the alternative processes considered in 2. Other oxidants can also be used.

4. By minimizing the organic precursors before any chlorine application is made (at the very end of treatment). This is a basic approach for raw waters of mediocre quality where both the precursor content and chlorine application may be substantial.

5. By carefully controlling the conditions of raw water transportation and potable water distribution, as these may be major sources of organochlorines in drinking water: (a) chlorination of *raw waters* (a neglected but frequently important organochlorine source) should be avoided and replaced by alternative approaches (clarification of water before transportation, mechanical cleaning, etc.). Using oligotrophic waters would favorably resolve the problem; (b) when and where a chlorine residual is judged necessary in the *distribution network*, it should be kept as low as possible; good bacteriostatic agents such as chloramines or chlorine dioxide may also be used. Good

maintenance and cleanliness of the networks are of great importance; they contribute to biological safety and help minimize the formation of organochlorines (through lower dosing of chlorine residual and lower organic content in the pipes).

Careful and moderate use of chlorine is not condemned, but more prudence and selectivity are required in its use as there is concern about operating practices that are not cognizant of the problems of byproducts and do not attempt to minimize them. Chemically and biolgically safe water remains the central goal of drinking water supplies.

CONCLUSIONS

1. Although organochlorinated substances may already be present as pollutants in raw waters, chlorination of water containing natural or synthetic organic precursors is generally by far the main source of halogenated organic chemicals in drinking water, as most surface waters contain substantial amounts of precursors. This is especially true of eutrophied waters which are generally very rich in organic substances.

2. Trihalomethanes (including chloroform) are currently the more easily identified organohalogen byproducts, but normally they represent only a modest proportion (about 20 percent) of the total organohalogens present in drinking water, and not necessarily the most hazardous substances. Although a large proportion of organohalogens present in drinking water are still unidentified, useful overall or partial parameters have been developed to permit a closer assessment of the presence or potential formation of organohalogens in drinking water. Total Organic Chlorine is the most relevant test as it is comprehensive and applies to the whole range of organochlorines present (however, it does not individually identify compounds). Total Organic Carbon is a useful complementary test for assessing the potential amounts of precursors. The Trihalomethane analysis is a relatively easy test but only gives a partial view of the total mix of chemicals present.

3. Oxidants such as chlorine, ozone, chlorine dioxide and to a lesser extent chloramines, are effective reagents in drinking water treatment, especially for disinfection, their essential function. However, being chemically very active, they may produce a variety of byproducts by reacting with the organic precursors present in waters. Up to now, organochlorinated byproducts have received most of the attention for a number of reasons: They are frequently encountered in significant levels in drinking water; a number of organochlorines are known or suspected to present health hazards, and they can currently be detected with present techniques. Although knowledge is very limited, it would be prudent to consider the possible effects of the byproducts which may arise from the use of other oxidants.

4. In many drinking water treatment installations, chlorine is used extensively throughout the system from the initial raw transportation to final drinking water distribution. It is clear that chlorine applications, particularly from the early stages when water may still contain substantial levels of organic precursors, will lead to significant organochlorine formation. A better control of organohalogens in drinking water requires more selectivity in the use of chlorine, which should, as far as possible, be kept to its essential role of final disinfection. For the bacteriostatic effect in the network, a chlorine dioxide or chloramine residual is an effective alternative.

5. In principle, processes which remove or reduce contaminants (physical and biological treatment) should be preferred to processes such as chemical treatment which transform them into other chemicals with undesirable or unknown effects. Authorities should also specify and control the quality of additive chemicals used in potable water treatment.

6. The gradual decrease frequently noted in the quality of raw waters used over the past few decades, has intensified treatment. The parallel increase of both organic pollutants in waters and chlorine applications all along the treatment system has lead to the organohalogen levels currently encountered in drinking waters. Using good quality raw waters is thus fundamental to controlling organohalogens and other trace pollutants in potable water.

7. Breakpoint chlorination, commonly practiced for ammonia removal, may lead to high levels of organochlorines in drinking water. Thus it seems advisable to use other ammonia removal methods such as biological removal, storage, resins, better protection of the source, or combinations of these processes. Under exceptional circumstances (e.g., during cold periods) when breakpoint chlorination is used, it should be carried out as a final disinfection treatment stage, after removal of organic precursors.

8. Under certain geographical and geological conditions, raw waters of good quality may, however, have a high content of humic and fulvic acids. Although these substances may not in themselves present a real hazard to human health, they react readily with chlorine to form organochlorinated compounds. Precautions should be taken with these waters so that the processes used throughout the water transportation, treatment and distribution system, minimize the formation of organohalogens. Similar caution is required with sources subject to seawater intrusion and bromide contamination, as chlorination will lead to the formation of significant amounts of both organobromides and organochlorides.

9. Where chlorine is used for purposes other than disinfection (e.g., keeping the treatment plant clean and free of biological growth) alternative approaches should be adopted (such as shock-dosing and rinsing of installations).

10. Chlorination of raw waters during transportation, carried out for secondary purposes only (control of fixed organisms) may be a very important source of organohalogens; however, it is generally underestimated or neglected because it does not take place in the plant. A number of alternative processes such as clarification of water before transportation, mechanical cleaning, shock-dosing and rinsing, etc., can be used successfully.

11. When a distribution system is in poor condition, high chlorine dosing is often used to maintain substantial disinfectant residual, and in the presence of

precursors the formation of organohalogens will continue as long as chlorine persists in the system. Good maintenance of the distribution network contributes to biological and chemical safety of water as it minimizes the use of a chlorine residual.

12. The microbiological quality of drinking water is, like its chemical quality, of prime importance, and biological safety should not be compromised when improving the chemical quality. Sufficient technologies are available to optimize both biological and chemical purity of water at a cost which, especially for larger water systems, is generally a modest fraction of the consumer's cost for drinking water. Thus, it is false economy to sacrifice drinking water quality by not applying optimal treatment.

13. The increased risk of cancer due to organochlorinated compounds in drinking water cannot yet be fully evaluated. Besides the estimates on health risks from chloroform, knowledge is still lacking on the potential hazards from the large number of unidentified organohalogenated compounds encountered in drinking water; they may be much higher. As there may be no "safe" level for these substances, it is justifiable to maintain their concentration at the lowest practical level.

14. It is desirable that guidelines be fixed, preferably in terms of total organic chlorine at the consumer's tap. Sufficient flexibility should be left in application, especially in different cases. However, unless the goals expressed in the guidelines or standards are stringent enough they may have a negative effect on a large number of water works which are already within the limits fixed, and act as a disincentive for any further improvement. In other words, they must not be the lowest common denominator but should be focused on the best levels realistically attainable.

APPENDIX

Precursors are natural or synthetic organic compounds capable of reacting with chlorine (and other halogens) and producing organochlorinated compounds (or more generally organohalogenated compounds).

Organohalogens (or organohalogenated compounds): Organic compounds whose molecule contains 1 or more halogens (such as chlorine, bromine and iodine).

Organochlorines (or organochlorinated compounds): Organic compounds whose molecule contains 1 or more chlorine atoms.

Volatile organohalogens: the molecule contains less than 4 atoms of carbon.

Non-volatile organohalogens: the molecule contains 4 or more carbon atoms.

Trihalomethanes (THMs) are volatile organohalogens whose molecule contains 1 atom of carbon, 1 atom of hydrogen and 3 atoms of halogen. When there are 3 atoms of chlorine it is **Chloroform**. When there are 3 atoms of bromine it is **Bromoform**. For instance, when there is 1 atom of bromine and 2 atoms of chlorine, it is monobromodichloromethane, etc.

THE IMPACT OF TOXIC TRACE ELEMENTS ON INLAND WATERS WITH EMPHASIS ON LEAD IN LAKE MICHIGAN

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ABSTRACT

Because of their widespread distribution and potentially significant detrimental effects, trace metals as pollutants have been extensively researched. Attention has focused on point sources and gross pollution, selected heavy metals, metal-organic associations and complexes, extremely toxic metals, transport mechanisms, metal species which undergo biological transformations, nonpoint sources, and sinks of trace metals. Non-metal, non-nutrient trace elements like As and Se have also become of concern. Fundamental understanding of the impact of trace elements on inland waters has not yet, however, been accomplished. Analytical and sampling limitations coupled with low concentrations encountered and possible multiplicity of species present have hampered progress. Also, the criteria for measuring impacts are not well developed. In particular, little is known of the biological effects of low level chronic exposures and potential synergistic effects. Some progress has been made. Considerable information is now available on common sources and sinks of trace metals. Modeling efforts and analytical developments are advancing toward fundamental predictive capabilities, and toxicological research has progressed beyond simple acute response measurements. Nonpoint sources, especially atmospheric inputs, particulate phase transport, sediment processes, and biochemical transformations are recognized as being critical to management strategies. Trace elements most likely to have adverse impacts have been identified, and strategies for impact assessment have been developed. Lead in the sediments of Lake Michigan offers a useful study example to review current knowledge and capabilities.

INTRODUCTION

Earlier considerations of trace metal pollution have focused mainly on metals in drinking water. Standards for lead, copper, and zinc, for example, were first set by the U.S. Public Health Service in 1925 (Pojasek, 1977). Progressively more complex problems have been approached as analytical capabilities have improved and environmental concerns have broadened. Increasing inputs of trace metals to natural waters from man's activities have been documented. Non-metal, non-nutrient trace elements, like As and Se, are also potentially significant pollutants.

Fundamental and comprehensive understanding of the impact of trace metals on inland waters has not, however, been accomplished. Analytical and sampling limitations, low concentrations, and numbers of species present have hampered progress (Stumm and Morgan, 1970; Stumm and Bilinski, 1973; Brewer and Spencer, 1975; Am. Chem. Soc. 1978). Sources and sinks of trace metals are relatively easily identified, but important species, cycling processes, and controls on concentrations are more difficult to determine. Also, the criteria for measuring impacts are not well developed. In particular, knowledge of the biological effects of low level chronic exposures and potential synergistic effects is incomplete.

Some progress has been made. Currently available knowledge can be used to improve investigative, management, and restorative practices. This paper reviews some current knowledge and shows its application to Lake Michigan.

ASSESSMENT OF TRACE METAL POLLUTION IN NATURAL WATERS

Complete understanding of the inputs and outflows, the physical, chemical, and biological forms and interactions of trace metals within the system, and the significance of changes in these characteristics, is the unattained goal of trace metal aquatic pollution research. Extensive and valuable information is available and has been reviewed in detail (Stumm and Morgan, 1970; Martell, 1971; Schnitzer and Kahn, 1972; Singer, 1973; Stumm and Baccini, 1978) Reviews of analytical techniques for trace metals, which point out the limitations of currently available data and the possibilities for advances based on new analytical approaches are also available (e.g. Mancy, 1971; Burrell, 1974; Quinby-Hunt, 1978). Pojasek (1977) and Ketchum (1972) present systems approaches to impact assessment, and James (1978) and Jenne (1979) have edited reviews of modeling approaches that are relevant to trace metal pollution.

Many problems, like metal speciation and its relation to toxicity, result directly from underdeveloped study techniques (Andrew, et al. 1976; Cantillo and Segar, 1975), but some shortcomings relate to broader concepts. Three generalizations summarize the problem: (1) Investigations are often too limited in scope to contribute to general advances — they focus on areas prescribed by current "fads"; (2) attempts to synthesize information have often become merely mathematical techniques; and (3) the importance of the relationship between data, regulations, and environmental impacts is often subjugated to other considerations. For example, a list of fads in aqueous trace metal research might include gross pollution from point sources, waste treatment processes, metal-organic associations and complexes, adsorption, biomethylations, and nonpoint sources. Evidence supporting the second generalization might include box models complete with arrows and labels for concentrations and transfer coefficients but with little indication of interest in obtaining that information. Finally, an example of the third generalization is the current emphasis on expedient effluent guidelines rather than receiving-water effects (Andrew, et al. 1976).

The point of these generalizations is not to be critical. Progress in investigations usually requires limiting their scope. Simplifications are necessary. The art of science is making the best simplifications. The point is to recognize each generalization as a step only and not as an end in itself, and to ask many and varied questions, including the difficult ones. Despite years of investigation and vast amounts of information, the integrative and predictive capabilities required for impact assessment and effective management of inland waters are not adequate.

What are current capabilities and how should impact assessment and management be approached? Numerous answers are possible and many different ones would have validity. The approach taken here does not claim to be original, nor to overcome the limitations mentioned. It is, rather, intended to represent some current approaches and knowledge.

Definition of the System

Impact assessment first requires delineating a scope of interest that attempts to include all relevant factors but remains manageable in size. For trace metal pollution, minimum consideration includes: (1) Organisms or ecological subsystem affected or to be protected; (2) elements and forms of elements added and present; and (3) physical locality. Note that the three areas of decision are not independent of each other. Essentially, definition of the system results from a combination of assumptions, previously available information, and perceived interests. Flexibility in changing the system must be maintained. The validity of the system chosen is critical to both the attainment and the usefulness of the results. It is impossible to answer all questions, so the goal must be to answer the right questions.

Organisms or ecological subsystems: Understandably, primary consideration of health effects normally centers on humans. However, the importance of broader concerns of environmental impact has been established and protection of many aquatic organisms is desired. In the case of Lake Michigan, no direct health effects on humans resulting from trace metal pollution of the water or fish are evident (Torrey, 1976; Andrew, et al. 1976; Int. Joint Comm. 1978, 1980). In fact, the offshore water of Lake Michigan meets all International Joint Commission target criteria for trace metals which also consider effects on aquatic organisms (Torrey, 1976; Int. Joint Comm., 1978; and Table 3). Hg in fish has been of concern in some of the lower lakes (Int. Joint Comm. 1978). Hg has received considerable attention in other systems, of course, including adverse effects on human health.

If all criteria are being met, should further consideration be abandoned? The answer is no. The fact that all criteria are being met can mean either we know all the answers, or we didn't ask the right questions. In the words of Brown (1976), "we might be in danger of outsmarting ourselves." The complicated nature of toxic reactions has become evident and sublethal effects (see Table 1) as well as acute effects must be considered. The common approach of setting standards for individual metals in aquatic systems overlooks possible additive, synergistic, and antagonistic effects as well as variations in external environmental factors and previous history of the organism (Zitko, 1976; Anderson and Weber, 1976; Cairns, et al. 1976). For example, International Joint Commission standards for individual metals were set below levels thought to have measurable toxicity to algae, but a mixture of all of the metals at concentrations just 10 percent of the standards proved toxic to test algae (Int. Joint Comm. 1978; Wong, et al. 1978).

Present abilities to predict or even measure toxicity, especially under environmental conditions, are improving but are still limited (Zitko, 1976; Dagani, 1980; Water Pollut. Control Fed. 1980; Am. Chem Soc. 1978). Research is being conducted on new measures of toxicity like ATP activity (Riedel and Christensen, 1979), low level *in situ* techniques (e.g. Marshall and Mellinger, 1978), multiple factor and synergistic toxicity (e.g. Anderson and Weber, 1976; Vernberg, 1978), and continuous sublethal monitoring (Dagani, 1980; Bruber, et al. 1979). However, in most cases management decisions must still be based on assumed or potential impacts, and impact assessments often rely on arbitrary safety factors rather than detailed knowledge.

Although younger life stages and some species are more susceptible than others, universally acceptable indicator species are not available (Dagani, 1980; McKim, 1977; Brown, 1976). Few impact assessments or management decisions will be afforded the luxury of limiting concern to one or two organisms, but as a practical matter, choices will have to be made.

Table 1. — Examples of sublethal effects (U.S. EPA, 1979).

Disruption of Normal Behavior (feeding, breeding, locomotion)
Interference with Thermoregulation in Birds and Mammals
Abnormal Biological Processes
Decrease in Reproductive Success
Change in Growth Rates
Effects on Competitive Balance and Predator-Prey Relationships
Shifts in Population Age Structure
Mutagenicity, Teratogenicity, and Carcinogenicity

Table 2. — (Brown, 1976; Wood, 1975; Ketchum, 1972).

Widespread Toxic Elements	Metals with High Anthropogenic Mobilization Rates	
	Element	Man-induced rate / natural rate
Co, Bi, Ni,		
Cu, Zn, Sb,	Sn	110.
Cd	Sb	31.
Widespread Toxic	Pb	13.
Elements Also Potentially Present	Fe	13.
as Metal-alkyls	Cu	12.
Sn, Se, Te,	Zn	11.
Pd, Ag, Pt,	Mo	4.4
Au, Hg, Tl,	Hg	2.3
Pb	Ag	1.4
	Ni	1.1

Elements and forms of elements: The impact of any trace metal input is a complex function of (1) the amount and timing of the input, (2) the form of the input and the forms present after any transformation in the water system, (3) the exposure of organisms to the input, and (4) the innate toxicity of forms present. Exposure is governed substantially by the physical characteristics of the water system relative to ecological habitats, and the nature of the input material is governed by the source. The forms present are controlled by the characteristics of the water system and the elements.

Of the 92 elements from hydrogen to uranium, all but 22 are metals. Several other relatively toxic elements, especially As and Se, have some metallic and some non-metallic characteristics which give them complex aqueous chemistries (Holm, et al. 1979). The term "trace" metal is a relative term that has exact meaning only in specific cases but generally infers a concentration below 1 mg/l (Brown, 1976).

Adverse impacts could result from adding sufficient amounts of any metal. A combination of high degree of toxicity, tendency to accumulate in organisms, and widespread distribution has been used to indicate potential hazards. Since trace metals are naturally ubiquitous and many are essential in biochemical processes, a further refinement in targeting potential impacts has been to use the ratio of anthropogenically mobilized metal relative to natural flux rates. Table 2 summarizes elements which appear as problems in analyses based on these criteria. Table 2 overlooks many factors, but eight elements, Sn, Sb, Pb, Cu, Zn, Hg, Ag, and Ni, appear on both lists and are, presumably, especially worthy of consideration. Studies conducted for the International Joint Commission have also developed a list of elements of concern: Pb, Cu, Zn, Hg, As, Se, Cd, Cr, and V (Int. Joint Comm. 1978, 1980). Commission objectives and example Lake Michigan concentrations for these elements are presented in Table 3.

The International Joint Commission objectives, like most standards set for trace metals, are based on total metal concentrations, with the exception of Hg, which pertains to filtered samples. As discussed in the trace metal aquatic chemistry reviews summarized in Table 4, numerous forms of any one element can exist in a natural water system. Toxicity is known to be a function of the specific forms present (Lee and Hoadley, 1967; Am. Chem. Soc. 1978; Andrew, et al. 1976). Some progress is being made in relating forms of metals in aquatic systems to toxicity and organism accumulation (Andrew, et al. 1976, 1977; Whitfield and Turner, 1979; Magnuson, et al. 1979; Vernberg, 1978). In many cases, the most toxic form of the metal seems to be the "free" cation (hydrated only), except that alkylated species, when present, are generally even more toxic than inorganic forms. The form of metal present probably affects potential accumulation in organisms as well as direct toxicity (e.g., Dodge and Theis, 1979).

On the basis of input rates, tendencies to be biomethylated, and potential accumulation in sediments and biota, the International Joint Commission considers Pb and Hg to be of greatest concern in the Great Lakes (Int. Joint Comm. 1978). Concentration objectives, however, are still given in relation to total concentrations (except for Hg). Current knowledge does not allow setting criteria for specific forms since

Table 3. — Offshore water concentrations and objectives for designated elements (Int. Joint Comm., 1978; Torrey, 1976; Elzerman and Armstrong, 1979).

Element	IJC Objective (μg/l)	Typical Offshore Lake Michigan Total Concentration (μg/l)
Hg	0.2	0.02-0.20
Pb	25.	0.8
Cr	50.	3.0
Cd	0.2	0.03
Cu	5.0	1.2
Zn	30.	1.2
Se	10.	0.1
As	50.	1.1

Table 4. — Summary of major forms of trace metals in aquatic addition to variations in oxidation state).

Dissolved	Non-Living Particulate	Living
1. "Free" Ion (hydrated only)	1. Adsorbed, as any of dissolved forms	1. Adsorbed
2. Inorganic complexes	2. Precipitated; Amorphous or crystalline, including substitutions and co-precipitates	2. Absorbed
3. Organic complexes		
4. Alkylated or other organic		

Table 5. — Lead distribution in southern Lake Michigan (Elzerman, 1976; Int. Joint Comm. 1978; Edgington and Robbins, 1976).

Reservoir	Typical Conc.	Reservoir Vol.	Total Mass	% of Total
Water: Dissolved	0.7 μg/l	1.5×10^{15}l	1.1×10^{15}g	85
particulate	0.1 μg/l	1.5×10^{15}l	0.2×10^{15}g	15
Biota: Plankton	?	Negligible	Negligible	Negligible
Fish	0.5 μg/l	Negligible	Negligible	Negligible
Sediments: top cm	120 μg/g	1.8×10^{14} cm^3	4.3×10^9	Negligible
1-4 cm	80 μg/g	5.4×10^{14}cm^3	1.7×10^{10}	Negligible

(porosity = 0.9; solids density = 2 g/cm^3)

TOTAL 1.3×10^{15}g

interconversion of forms is an insufficiently understood possibility. Distinction between dissolved and particulate forms and careful control of ambient parameters are minimum requirements for future investigations. A general consideration of trace metal impacts must, at this stage, still consider total concentrations.

Physical limits of the system: The physical limits of the system chosen for consideration cannot be arbitrary. The boundaries delineate the sources, sinks, and internal processes of the system. Judicious choice of boundaries can greatly simplify a study. For example, if As from a point source in a harbor is to be considered (e.g. see Holm, et al. 1979), a logical choice of boundaries may be the sides of the harbor and the outlet flow to the point where the As concentration is diluted to background lake water levels. In some cases, it might make more sense to follow the system to where sediment concentrations decrease to normal regional values. Partial physical barriers often make convenient boundaries; Lake Michigan is often divided into a southern more industrialized and a northern less industrialized basin by the submerged ridges running between Milwaukee and Grand Haven.

Collection and Analysis of Information

After the scope of an impact assessment has been defined, the next task is accumulating needed information. Obtaining as much relevant information as possible and abstracting the most useful is frequently beneficial. Already available information may need to be supplemented by investigation to obtain new information. All information must be analyzed for quality and significance to determine its importance to the assumed goals of the assessment.

Distribution in the system: Knowing the distribution of a trace metal in different reservoirs can be useful. Representative values for southern Lake Michigan are given in Table 5. Note that values given are estimates of variable reliability and the original references should be consulted for further information. Assumptions made ignore the higher lead concentrations in nearshore waters and the uneven distribution of lead in the sediments (Edgington and Robbins, 1976; Int. Joint Comm. 1978). Although crude, the estimates indicate the water column is the most significant reservoir in terms of mass of lead. High concentrations of lead are found in the sediments and significant amounts are buried below the top active zone (here taken as the top 4 centimeters), but the overlying water actually contains more lead. Little information on Pb levels in Lake Michigan biota, especially plankton and bacteria, is available. Pb bioaccumulation factors are generally in the range of 10^2 to 10^3 (Ketchum, 1972; Callahan, et al. 1979). The relatively insignificant mass of biota makes it a negligible reservoir. Similarly, large concentrations of lead can be found in the surface microlayer (Elzerman and Armstrong, 1979), but its relatively small volume makes the mass of lead in this reservoir insignificant.

Of course, the fact that a reservoir contains a small fraction of the total lead does not mean lead interactions are insignificant. For example, alkylation of lead has been observed (Wong, et al. 1975; Chau, et al. 1979), but whether it occurs primarily in the sediments, water column, or organisms is unknown. Metal alkylation in the environment is known to be a complex process (Ridley, et al. 1977).

Forms present: Information on thermodynamically expected forms for most trace metals in aqueous systems is now readily available and , although subject to the limitations of thermodynamic predictions, very useful. Numerous computerized models, like REDEQL, MINEQL, and GEOCHEM, are widely used for species prediction (Jenne, 1979), and summaries of metal speciation are available (e.g. Callahan, et al. 1979). Kinetic controls on concentrations and the nature of particulate phases present are not as easily predicted.

Soluble forms of Pb are influenced greatly by pH and the anions present, especially carbonate (Callahan, et al. 1979; Davis, 1976). The free ion (Pb^{+2}) dominates only at low pH. At intermediate pH's, species such as $PbCO_3^0$, $PbOH^+$, and $Pb(OH)_2^0$ are important. Sufficient Cl^- of SO_4^{-2} can lead to the presence of $PbCl^+$ and $PbSO_4^0$. . Pb appears to be strongly complexed by organic materials and readily adsorbed by particles in natural waters. Particulate lead in the water column is probably mostly adsorbed or in organisms. Elzerman, et al. (1979) found evidence of significant fluxes of high Pb concentration ($>$5,000 μg/g) atmospheric particles to the lake and almost all Pb in the surface microlayer to be in particulate form. More recent evidence (Elzerman, et al. 1980) suggests that some of the Pb quickly dissolves from the aerosol in the lake water and may then be readsorbed by particles in the surface microlayer. In sediments, $PbSO_4$ (PbS if S^{-2} is present), $PbCO_3$, and complexed or adsorbed Pb are likely solid components.

Boundary fluxes: An important consideration, especially to contemplated management measures, is the sources and sinks of a trace element to the system. The major source of Pb to Lake Michigan seems to be the atmosphere (Edgington and Robbins, 1976; Cogley, 1974; Eisenreich, 1980). Atmospheric inputs to Lake Michigan have been extensively studied (e.g. Winchester and Nifong, 1971; Klein, 1975; Gatz, 1975; Eisenreich, 1980, summarized in Table 6). Estimates of the fraction of the total Pb input attributable to the atmosphere range from 60 to almost 100 percent. International Joint Commission estimates (1978) of Pb inputs to the whole lake are 190 metric tons per year from point sources and 1,670 metric tons per year from nonpoint sources (including the atmosphere). The only substantial sink of Pb from the system is the sediments, where it seems to be essentially immobile (Edgington and Robbins, 1976; Cogley, 1974).

Mass balances and residence times can be estimated from boundary fluxes and reservoir loadings (Bowen, 1975). For example, Edgington and Robbins (1976) estimated the flux of Pb to southern Lake Michigan in 1972 to be 270 metric tons per year (240 from the atmosphere). Approximately the same flux to the sediments was found. A simple calculation for the water column based on this flux and the mass of Pb in the water column reservoir (Table 5) indicates a residence time of many thousands of years, but processes like transport to the sediments in particles make the actual residence time much less (Brewer and Spencer, 1975). As a result, increases in Pb inputs to the lake are exhibited as increased concentrations of Pb in recent fine-grained sediments of active depositional regions (Int. Joint Comm. 1978; Edgington and Robbins, 1976; Leland, et al. 1973).

Table 6. — Trace metal inputs to Lake Michigan (Eisenreich, 1980.)

Element	Tributaries	Shore Erosion	Atmosphere	% Atmsopheric
		10^3 kg/yr		
Pb	180	240	640	60.
Zn	500	1,800	1,100	32.
Ca	18,400	280,000	79,800	21.
Cu	230	540	120	13.
Mn	850	4,100	640	11.
Cd	12	75	11	11.
Fe	36,000	2,300	2,770	6.7
Mg	8,800	250,000	15,500	5.7
Al	17,500	75,000	4,990	5.1
Co	15	700	25	3.3

Internal cycling and transport: Biological, chemical, and physical transformations within an aquatic system are numerous, complex, and not well understood, especially in relation to the rates at which they occur (see Stumm and Baccini, 1978, for review). Physical transport mechanisms, including advection, dispersion, and diffusion, have been more successfully described and probably account for the major horizontal movements of Pb within the lake. Vertical dispersion coefficients tend to be much smaller than horizontal dispersion coefficients (Thibodeaux, 1979); therefore the major vertical movement of Pb probably results from sorption by particles or organisms followed by sinking (Brewer and Hoa, 1979; Ferranti and Parker, 1977; Brewer and Spencer, 1975). Internal cycling and transformation affect the forms of Pb present, the exposure of organisms to Pb, and the removal of Pb from the system. Leland, et al. (1973) have reviewed factors controlling the high concentrations of Pb in sediments.

Integration of Information and Objectives

Regardless of the quality and quantity of information accumulated, final integration of information and objectives is necessary, often difficult, and likely to require non-scientific decisions. Frequently, substantial modeling efforts have been undertaken to improve interpretation and implementation of results. Modeling has not always been successful, especially for comprehensive problems, but useful approaches like EXAMS (Lassiter, et al. 1979) and transfer models (e.g. Wiersma, 1979) have developed. Models are probably most useful for sensitivity analyses (estimating responses to various perturbations) and as part of a general systems approach (Pojasek, 1977; Ketchum, 1972). Throughout the investigation, and particularly when arriving at conclusions, the applicability of the data to intended uses must be reviewed. For example, information presented on Pb in Lake Michigan does not define the impact of alkylated Pb compounds on Lake Michigan fish, but it can be used in predictive models to evaluate different remedial measures to control Pb inputs to the lake. Control of atmospheric inputs would be most significant, but difficult to achieve. Heidtke, et al. (1980) have shown that control of rural and urban runoff sources would be expensive and only of limited usefulness. Consequently, lead inputs to the lake are likely to continue at significant levels and the need for further assessment of the fate and effects of Pb in the lake remains.

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WATERBORNE GIARDIASIS

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ABSTRACT

Waterborne disease outbreaks occur in the United States at an average rate of 35 per year and have been increasing since 1950. Giardiasis is a waterborne illness of major concern and accounted for 36 outbreaks affecting 16,000 people in the past 15 years. The number of cases is thought to be considerably underestimated. The most serious outbreaks have occurred in communities using reservoirs as a source of water supply. Water systems using reservoirs as a source of supply depend on long storage time and environmental factors to reduce turbidity and permit microbiological die-off to occur. Therefore, minimum treatment is provided and in many cases chlorination is the only protection. *Giardia* cysts, unlike other pathogenic agents, can survive for long periods of time in the water environment and overcome the natural barriers provided by reservoir storage. They are also more resistant to chlorination. Chlorine dosage must be increased to inactivate cysts, creating the undesirable side effect of producing additional trihalomethanes. Increased chlorination provides a solution for an acute disease outbreak but may contribute to a chronic health problem.

BACKGROUND

Waterborne disease outbreaks in the United States have been recorded in the literature since 1920. For an historical perspective, the annual number of outbreaks based on averaging data for 5-year periods, is shown in Figure 1. A peak occurred during 1936-45 and it appears that we are approaching that peak in the current 5-year period. The trend has been increasing since the 1950's and has caused some concern. The largest outbreak recorded occurred in 1926 in Detroit, Mich. and affected 45,000 to 50,000 people with acute gastrointestinal illness. The most recent large outbreak occurred this year in Texas and affected over 8,000 people.

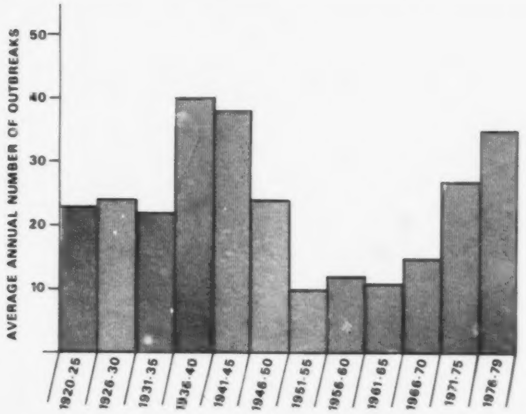

Figure 1. — Waterborne disease outbreaks, United States 1920-79.

Waterborne giardiasis outbreaks are relatively new in this country with the first reported in 1965. A total of 36 have been reported through 1979, as shown in Table 1. Two large outbreaks not included in the table are now thought to be waterborne giardiasis. One occurred in Portland, Ore. in 1954-55 affecting an estimated 50,000 persons and the other in Boulder, Colo. in 1972 with 300 cases. The Portland outbreak occurred at a time when there was considerable doubt about the pathogenicity of *Giardia* even though it was detected in stools of those experiencing symptoms that are now known to be typical of the illness. The Boulder outbreak was termed inconclusive because the organism was not identified in the water system and young adults were the group mainly affected, contradicting a belief that the high risk group was children. In approximately 55 percent of the reported waterborne outbreaks, the causative agent is not determined so the true incidence of waterborne giardiasis outbreaks could be considerably more than the number shown in Table 1. There is also general agreement that many outbreaks occur that are not investigated and consequently not reported.

Table 1. — Waterborne giardiasis outbreaks U.S. (1965-1979)

Period	Outbreaks	Cases
1965-69	2	142
1970-74	12	361
1975-79	22	15,407*
TOTAL	36	15,910

*4,000 cases estimated for preliminary 1979 data

The location of giardiasis outbreaks is shown in Table 2. They occur predominantly in the mountainous areas of the country, particularly in the Rocky Mountains, New England, and the Pacific Northwest. Many of these areas depend on sources of water supply that are not influenced by wastewater discharges and, consequently, minimal water treatment measures are employed. In many cases, chlorination is the only treatment used. Chlorination, as presently practiced by most water utilities, is not effective in inactivating *Giardia* cysts.

Table 2. — Location of waterborne outbreaks of giardiasis, U.S. (1965-1979).

State	Outbreaks/ State	Total
Colorado	11	11
Utah	4	4
New Hampshire, New York, Oregon	3	9
California, Montana, Vermont, Washington	2	8
Arizona, Idaho, Pennsylvania, Tennessee	1	4
		36

THE ORGANISM AND THE DISEASE

Giardia is a single-celled protozoan organism with two distinct stages in its life cycle. In a human or animal host it exists in an active or reproductive stage, termed a trophozoite. Outside the host it exists in an inactive or cyst stage. The cycle of infection for a human begins when the cyst is ingested either through contaminated food or water. When the cyst enters the upper small intestine, it excysts to the trophozoite stage and attaches to the epithelial lining where reproduction by binary fission, or splitting, occurs. Attachment to the lining of the small intestine apparently interferes with the digestive process, causing watery diarrhea, bloating, abdominal pain, and cramps. Eventually, trophozoites detach from the lining and begin to encyst in the small intestine and are excreted in the feces in the cyst stage. This is the form in which they are usually found in the feces; however, in some cases of severe watery diarrhea they are identified in the trophozoite stage. In the cyst stage they can survive for long periods in the water environment and have been reported as surviving for more than 3 months.

Feeding studies in the early 1950's determined the number of cysts required to produce an infection in humans. Prisoners who volunteered for the study were given cysts in their drinking water and it was found that 10 cysts were sufficient to cause infection. It is of interest to note that a person who is infected will shed an average of 15×10^6 cysts per gram of feces. A normal human stool weighs about 150 grams so the potential for one carrier in transmitting the disease is tremendous. (This translates to a capability of one person being able to contaminate a 50 mg reservoir to an infectious dose of 10 cysts/l.)

In this country, the illness is treated with three drugs. Quinacrine is normally the drug of choice and has a cure rate of about 95 percent.

PROBLEMS RELATED TO WATER SUPPLY

Twenty-nine of the 36 waterborne outbreaks of giardiasis were related to using inadequately treated surface water. As previously noted, the outbreaks occur where sources of water supply are not influenced by wastewater discharges and this explains the minimal treatment. In most cases, the communities relied on reservoirs for raw storage to permit natural forces to reduce turbidity and microbial populations. Treatment consisted of chlorination to destroy bacteria so the systems complied with drinking water standards for coliform bacteria.

Until 1975, not much attention was paid to waterborne outbreaks of giardiasis. During that year a large outbreak affecting nearly 5,000 people occurred in Rome, N.Y. Prior to that time 14 outbreaks in small water systems had affected about 500 people. The Rome outbreak was notable not only because it affected so many people, but because it lasted for 6 months. There were no wastewater discharges in the watershed; however, a few malfunctioning septic tanks were discovered in the 200 square mile drainage area. Only four of 257 samples collected from the distribution system during the outbreak showed evidence of coliform contamination.

One year later, an outbreak at Camas, Wash. affected 600 people. This outbreak was especially notable in that *Giardia* cysts were, for the first time, easily identified in raw and finished water, and the organism was found in beaver living near the water intake. In the past 2 years, beaver have been implicated in 8 of 12 outbreaks by identifying cysts in beaver feces, or by necropsy of animals trapped from the watershed. It has also been determined through feeding experiments that cysts isolated from beaver feces can infect humans and, conversely, cysts from human feces can infect beaver.

The management concerns that have developed because of waterborne outbreaks of giardiasis included control of beaver in reservoirs and watersheds especially where a water supply may be affected, sampling and laboratory methodology to identify the cyst and determine whether it is still viable, and treatment technology to remove and/or inactivate the cyst.

Control of Beaver

Controlling beaver in reservoirs and watersheds may be difficult, depending on how it is done. It also cannot or need not be applied in every situation. It obviously cannot be applied in large watersheds because of the cost and logistical requirements, and beavers need not be controlled in locations where water treatment facilities are adequate. Control is considered to be a viable alternative in relatively small watersheds and reservoirs, especially where water treatment is marginal.

Control does not imply destroying the animal. Where this is the method used, trouble can be expected from an aroused public. A more acceptable method is to live-trap the animals and relocate them where the impact of water quality is minimized. If suitable holding facilities are available, it would be desirable to hold them in captivity until the infection can be cured. Other measures include replacing deciduous trees around reservoirs with coniferous species, thereby removing the beavers' food supply and discouraging habitation.

It is surprising how upset people become at the thought of dead-trapping or sacrificing a few beaver. They somehow have perceived the beaver as a harmless animal rather than a member of the rodent family which can be quite destructive in a reservoir and around waterways. They are indiscriminate in selecting trees for felling and can create turbidity and trash problems in reservoirs. It is not difficult to determine whether beaver are present in reservoirs as stripped cuttings are usually piled up against spillways, dam faces, and around outlet structures.

Sampling and Laboratory Methodology

Regulatory agencies and water utilities have for years relied on coliform bacteria as indicators of drinking water quality and safety. Absence of the coliform group from drinking water normally indicates the water is free of pathogenic or disease-causing bacteria and virus. Laboratory tests for coliforms are relatively simple, inexpensive, and universally accepted. To produce water to meet the coliform standard, simple chlorination is used to effectively reduce bacterial counts to acceptable levels. In practice, 30 minutes' contact time is normally needed for chlorine to react with and destroy coliform bacteria; by current standards the water then is safe to drink. For the concentration of chlorine and contact time usually employed to achieve compliance with the coliform standards, disease-causing bacteria (*Shigella, Salmonella,* enterotoxigenic *E. coli*) and virus (Polio, Coxsackie, ECHO) are destroyed or inactivated. Therein lies the paradox.

Giardia cysts are much more resistant to chlorine than the coliform group of bacteria. They are not inactivated at concentrations and contact times used by many water systems. Traditional monitoring of drinking water for coliforms as required by law will indicate that the water is safe when it may be contaminated with *Giardia* cysts.

Current sampling methodology for cysts requires filtering large volumes of water (2,000 liters) through an orlon fiber filter tube to trap the cysts. Cysts are then removed from the tube through a laborious laboratory procedure which requires cutting the fibers from the tube and stirring the mass in a blender. The fluid expressed from this procedure is taken through a flocculation process to separate the organic and inorganic particulate matter also trapped on the filter, to obtain a suspension hopefully containing the cysts. The suspension is centrifuged and a few drops of the centrifugate are examined microscopically for the presence of cysts. Besides being time-consuming, the method is only about 6 percent efficient. Because of its inefficiency, a negative finding does not indicate absence of cysts, so its use in monitoring a water system is somewhat limited. The methodology is quite difficult and requires personnel with specialized training who are not normally available to laboratories that conduct routine analyses for water utilities. It also has the inherent disadvantage of not being able to determine viability of cysts viewed under the microscope. This has important implications related to water treatment.

Cyst viability or ability to produce infection is determined by feeding the flocculated suspension obtained in the laboratory to specific pathogen-free beagle puppies. A positive test is development of an infection in the pups which requires about 7 to 10 days. Only one location in the United States has the capability to conduct the feeding experiments. Needless to say, EPA is involved in extensive research to address monitoring and laboratory methodology problems.

Water Treatment and Control Technology

The multiple barrier concept which requires placing protective systems between the water consumer and actual as well as potential sources of contamination is of primary importance to insure the delivery of safe drinking water. Reservoirs play an integral part in the multiple barrier concept as they aid in reducing turbidity and microbial populations to improve water quality and assure a dependable supply of water during low flow periods. These assets are perhaps used to an unfair advantage by communities that rely on chlorination as the only means of treatment, especially so where *Giardia* contamination is a potential threat.

Conventional water treatment including chemical coagulation and filtration is effective in removing *Giardia* cysts from water and should be employed as an additional barrier where the surface water and reservoirs are used as a source of supply. Where surface water without an impounding reservoir is used as a source of supply, communities have had to install filtration facilities to reduce turbidity and produce drinking water of acceptable quality. As previously mentioned, however, outbreaks of giardiasis have occurred in areas where water supply sources are not influenced by wastewater discharges and raw water quality is better than average. In these areas, dependence has been placed solely upon reservoirs and treatment by chlorination which are not adequate barriers during *Giardia* cyst challenge.

Where outbreaks have occurred under these conditions, two emergency measures are implemented. A boil water order is issued and chlorination is increased to inactivate the cyst. Boiling water for 1 minute destroys the cyst but the extent to which the community complies with the order is not generally known. With energy costs now an important consideration in the family budget, people resent the added cost and burden of boiling their water. Increasing chlorination to destroy the cysts produces objectionable side effects of creating an unaccustomed taste and odor problem and may contribute to the building of trihalomethane concentrations. EPA has promulgated regulations to control trihalomethanes because of their cancer-causing potential, so by increasing chlorination to control an acute health problem, a potential chronic health situation may be created.

The long term solution is to fully implement the multiple barrier concept and provide adequate treatment with a desirable adjunct of controlling the beaver population in the watershed where it is practical.

Adequate treatment includes conventional unit operations capable of removing cysts, and disinfection at an acceptable concentration over a sufficient period of time for inactivation to occur, as an added measure of protection. The State of Colorado adopted regulations in 1977 requiring communities using surface water as a source of supply to provide treatment to remove *Giardia* cysts. Other States have similar regulations under consideration.

SUMMARY

Outbreaks of giardiasis are increasing in frequency and severity and occur predominantly in communities using surface water as a source of water supply. The most serious outbreaks have occurred in communities depending on reservoirs, with minimal treatment facilities as barriers of protection. While these barriers are adequate to produce water complying with the coliform standard, they are inadequate under *Giardia* cyst challenge.

Beaver have been increasingly associated in outbreaks as carriers of *Giardia* cysts. Feeding studies have shown that the cyst infecting beaver also infects humans and the converse is true. Control of beaver is appropriate and should be incorporated into the multiple barrier concept in certain situations.

There are problems in monitoring and laboratory methodology which require additional research. Improved sampling techniques, laboratory processing of samples, and a methodology for determining cyst viability are required. EPA is addressing these needs through in-house studies and research grants and contracts.

Water treatment technology is available to reduce and inactivate *Giardia* cysts and this technology should be applied to prevent outbreaks of giardiasis.

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RESIDENTIAL WELL WATER QUALITY IN WISCONSIN INLAND LAKE COMMUNITIES

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ABSTRACT

Older inland lake communities in Wisconsin are more likely than many areas to have degraded water supplies. Many homes sit on sandy soils with high water tables, have shallow wells close to their own or a neighbor's septic system, and may not comply with State sanitation or well codes. Concern over this condition led to an investigation of groundwater quality at two lakefront residences suspected of having failing septic tank systems. A statewide Extension education program for lake communities was also created which includes screening tests of home drinking water for at least coliform bacteria, nitrate-nitrite-N, and chlorides. Test results suggest that about half of the lake or river community wells tested appear to be contaminated to some degree and that better residential well water management is needed.

BACKGROUND

Inland lake communities (particularly older ones) in the Great Lakes Region comprise a relatively high risk area for degraded drinking water supplies. Soils in these areas are often sandy with a rapid rate of groundwater movement and a high water table. Wells frequently shallow, driven sand points; and the on-site waste disposal system aging and inadequate. In addition, the popularity of lake property leads to small lots and crowded conditions around the lake. These circumstances, combined with possible non-compliance with State well and sanitation codes, could contaminate the shallow groundwater and create a health risk when it is tapped by residential wells.

The data presented in this paper are derived from two sources: a 1979 lake research project, and a University of Wisconsin-Extension information program initiated in the summer of 1978. The program involves a screening test of residential well water samples collected by inland lake homeowners. Tests conducted include total coliform, chlorides, and nitrate-nitrite-N. The samples are analyzed at local university facilities and test results returned to the participants at a community meeting. At the meeting the parameters tested are explained, local hydrologic relationships discussed, and advice provided for the protection and use of the groundwater resource. Residents whose samples indicate unusually high chlorides or nitrate-N or have coliform or general bacterial colonies are advised to have their water further analyzed by a certified laboratory and to determine whether their water systems meet the minimum standards of the State well code.

PREVIOUS STUDIES

Ellis (1971) and Ellis and Childs (1973) demonstrated the groundwater intrusion and lateral movement of septic system effluent at Gull Lake and at Houghton Lake, Mich. This point was also made by Brandes (1975). However, these studies were primarily based on the measurement of nutrients and other chemical constituents of effluent. Brandes did observe fecal coliform movement up to 17 meters from drainfields and Mack (1972) reported the transport of both polio viruses and coliform bacteria from a restaurant drainfield to its well water supply 300 feet away. (This transport distance may have been facilitated by the fractured limestone underlying the study area.) In 1979, the Office of Inland Lake Renewal, Wisconsin, Department of Natural Resources, applied the Ellis and Childs study design to a series of residences on a central Wisconsin lake (Knauer, 1980). At two of these sites, bacterial samples were taken from a series of shallow monitoring wells placed in a line between the septic system disposal field and the lakeshore. At both sites, the number of colonies per 100 ml in the groundwater increased and peaked between the septic tanks and the lakeshore. In both instances, and for all parameters measured, these peaks were noticably greater than the background level measured in a control well located upgradient from the septic tank on each property. Parameters measured were: total coliform, fecal coliform, fecal streptococcus, and *Pseudomonas aeruginosa*.

While this cursory investigation and the references cited certainly do not indict on-site waste disposal techniques, they do raise the issue of possible wellwater contamination from this source, particularly in lake communities or areas of shallow groundwater, poorly constructed wells, and sandy soils.

INITIAL STUDY OF BACTERIAL MOVEMENT IN GROUNDWATER FROM SEPTIC TANK SYSTEMS

The Ellis and Childs technique was adapted to a lake community in south central Wisconsin. A series of three variable depth monitoring wells was installed at roughly 1/3 distance intervals between the septic tank system and the lakeshore at two homesites suspected of having defective septic tank systems. The soils at both sites are of mixed glacial origin, but are mostly sandy clay. Each sampling site consisted of three adjacent wells set 15 centimeters, 60 centimeters, and 120 centimeters below the normal groundwater table. Figure 1 illustrates the placement of the wells. The primary intent of sampling these wells for bacteria was to determine if such contamination could be demonstrated for a Wisconsin lake community.

Sampling was accomplished by first pumping each well to waste, waiting 3 hours, and then pumping out the sample, using sterile tubing. A sterile collection bottle was inserted in the line, and suction provided by hand pump. The tube was lowered to well bottom and then withdrawn a few inches to reduce sediment intake. Fresh sterile equipment was used at each well site. These precautions were, however, compromised to some extent since other investigators had used the same wells to collect chemical water quality samples.

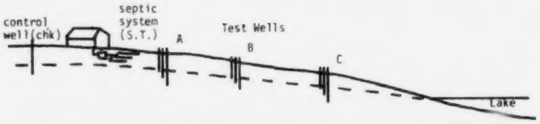

Figure 1. — Schematic representation of the test well system installed at two lake front residences on a lake in south central Wisconsin. Each cluster of three test wells (A, B, C) consists of a well set at the normal water table level (shallow), one set 2 feet deeper (middle), and one 4 feet deeper (deep). The control well (ch'k) sampled was equivalent to the middle depth.

Immediately after collection, the samples were returned to the laboratory for analysis. All analyses were by the most probable number (MPN) technique, in accordance with Standard Methods for the Analysis of Water and Wastewater (Am. Pub. Health Assoc., 1975). The results of the investigation are presented in Figure 2.

At both sites, the bacterial parameters in all cases increased between the drainfield and downgradient lakeshore. These increases consistently exceeded background levels as indicated by counts taken from a control well at each site located upgradient from the septic tank system and test wells. Later that summer attenuated Type I polio virus was introduced to the septic tank at site 2 as a tracer. The virus was later recovered from all of the test wells at the site and from the adjacent lake water and sediments (Stramer, 1980). Household water supplies apparently were not threatened by either of these septic systems since their wells were located elsewhere on the property, but the groundwater was being contaminated locally and the coliform, streptococcus, and *Pseudomonas* organisms remained alive and culturable at distances up to and exceeding 30 meters downgradient from the drainfields.

DATA GATHERED FROM THE EXTENSION INFORMATION DRINKING WATER PROGRAMS

While this study was admittedly rudimentary in nature, it was concluded that this evidence in conjunction with the body of existing literature was sufficient to justify further development and expansion of a pilot drinking water information program for Wisconsin lake communities. An assessment of data gathered from testing the residential drinking water in these communities would itself support or contradict the presumption of contamination risks peculiar to lake settings. Subsequently, 351 well water screening tests of residential well water systems have been conducted in 15 lake and river communities in Wisconsin (Figure 3).

The samples are analyzed for total coliform bacteria, nitrate-nitrite-N, chloride, and (variably) pH, specific conductance, hardness, and iron, in accordance with Standard Methods for the Examination of Water and Wastewater (Am. Pub. Health Assoc., 1975). In field settings, a portable kit augments standard laboratory techniques. All coliform analyses are conducted on 100 milliliter samples using the membrane filtration technique employing Millipore Corp. apparatus and disposable 5 centimeter diameter petri dishes and filters. Sample bottles are sterilized by autoclaving; field equipment is either packaged and sterilized by autoclave, or sterilized for reuse at the site by ultraviolet irradiation. The culture medium used is MF Endo Medium (BBL). All coliform test samples are filtered and incubated within 8 hours from time of collection. Confirmation is based upon the presence of metallic sheen colonies observed when the incubated cultures are read at 24 and 36 hours. Chemical tests are completed wihtin a maximum of 48 hours. Blanks and known reference samples are run with each set of chemical tests, and sterile water blanks and a positive control sample of contaminated water with each set of bacterial tests cultured. Usually no dilutions or replicates are made.

Water quality data compiled for the 15 programs are presented in Table 1. Total coliform colonies in the tests ranged from negative to "too numerous to count." Cultures which obviously generated colonial growth, but were lacking the classical metallic sheen were reported as atypical and are shown in column five. As a further indication of possible health risk, nitrate-nitrite nitrogen concentrations greater than 10 mg/l (EPA criteria re: risk of methemoglobinemia in infants) are shown in the next column. Chloride concentrations greater than background levels in the sample community are a possible tracer indicating effluent in the groundwater; they are also shown. Common sources of high chlorides are sewage, especially septic tank effluents containing water softener back flushes, road salt runoff and infiltration, animal waste leaching, fertilizer leaching, and natural salt deposits. The final column of total suspect samples is a summation of those water samples which exceeded the levels for any or more of the constitutents as shown in the table.

Values and percentages are not cumulative across the table because a given sample may reveal more than one suspect characteristics.

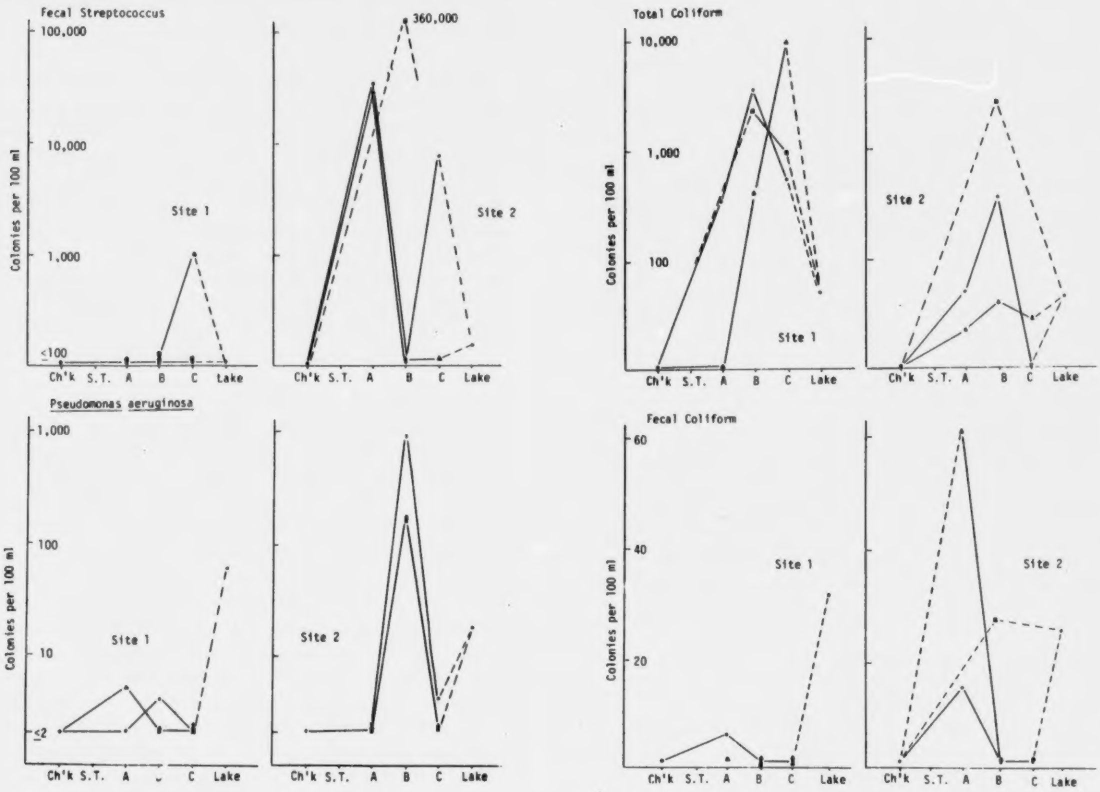

Figure 2. — Graphic presentation of the data collected from a single sampling of groundwater test wells at two lake front residences in south central Wisconsin. Bacteria concentrations are shown on the vertical axes, scales are not consistent. The horizontal axes repeat the linear arrangement of the test wells from the up-gradient control (ch'k) well, past the septic tank system (S.T.) to three equally spaced monitoring well clusters (A, B, C) each of which consists of three separate depth wells — shallow at the normal water table depth; medium 2 feet deeper; and deep 2 more feet deep. "Lake" indicates a sample taken at the lake edge in line with the wells. X = 1 data from shallow wells, ● = data from medium wells and also the control and lake observations, ▲ = 1 data from deep wells. Dashed lines indicate missing samples because the shallow wells were often dry. Dashes were also used between groundwater data and lake data. The trend is for bacteria concentrations to increase down gradient from the septic tank systems.

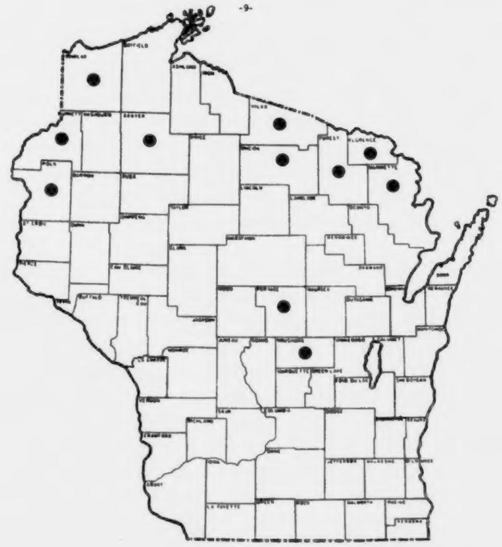

Figure 3. — Wisconsin counties in which Extension drinking water quality information programs have been held, 1978-1980.

Table 1: Residential well water quality data collected in fifteen Wisconsin lake or river communities from summer, 1978 to and including summer, 1980. The number of suspect water samples is less than the sum of values for each row because a given sample may be suspicious for more than one parameter.

Lake Community	N	No. of samples (100 ml) with "total coliform" colonies	No. of samples with "TC" colonies greater than one (EPA Drinking Water Criteria)	No. of samples with bact. growth but not identified as "TC"	No. of samples with NO_2-NO_3-N concentration greater than 10 mg/l	No. of samples with Cl concentration noticeably greater than local background levels (usually 10 mg/l	Total Suspect Samples
A	24	2 (8%)	1 (4%)	5 (21%)	1 (4%)	4 (17%)	8 (33%)
B	14	3 (21%)	2 (14%)	2 (14%)	0	4 (29%)	8 (57%)
C	26	4 (15%)	2 (8%)	14 (54%)	0	6 (23%)	17 (65%)
D	34	5 (15%)	3 (9%)	12 (35%)	1 (3%)	3 (9%)	16 (47%)
E	32	3 (9%)	1 (3%)	14 (44%)	0	4 (12%)	19 (59%)
F	29	9 (31%)	6 (20%)	Not reported	0	17 (59%)	22 (76%)
G	14	2 (14%)	1 (7%)	1 (7%)	0	1 (7%)	3 (21%)
H	32	7 (22%)	5 (16%)	12 (37%)	1 (3%)	9 (28%)	19 (59%)
I	27	1 (4%)	1 (4%)	3 (11%)	0	2 (7%)	6 (22%)
J	6	1 (17%)	0	1 (17%)	2 (33%)	2 (33%)	4 (66%)
K	24	4 (17%)	2 (8%)	14 (58%)	2 (8%)	3 (13%)	18 (75%)
L	26	11 (42%)	7 (27%)	22 (85%)	2 (8%)	6 (23%)	23 (88%)
M	27	3 (11%)	3 (11%)	16 (59%)	0	3 (11%)	17 (63%)
N	16	8 (50%)	6 (38%)	13 (81%)	0	1 (6%)	15 (94%)
O	20	0	0	2 (10%)	0	3 (15%)	5 (25%)
	351	63 (18%)	40 (11%)	131 (37%)	9 (2%)	68 (19%)	200 (57%)

DISCUSSION

Some of the variations involved in this approach include: samples are collected by the householders themselves; they are not instructed to flame the faucet before collection, but do purge the line; considerably less time elapses between collection of the sample and culturing than when samples are mailed to a laboratory; and incubation time is a minimum of 24 hours with a second inspection of the culture plates again at 36 hours. This additional 12 hours of incubation was elected because experience has shown that small colonies may be missed when the plates are incubated for only 24 hours. Colonial development after the initial 24 hour incubation may be stressed coliform bacteria or some other form, such as *Pseudomonas* or *Aeromonas*. It has been practical so far to verify such subsequent results.

The likely effect of these variations is to produce a greater frequency of coliform and/or atypical colonial growth than might be reported by conventional sampling and laboratory methods. When the multiple tube dilution technique is employed by a lab, further deviation might occur because most laboratories do not confirm non-gas forming, but nonetheless cloudy fermentation tubes which might be masking coliform occurrences.

This entire question of atypical colonial growth remains to be addressed. The EPA Microbiological Methods for Monitoring the Environment (Bordner, et al, 1978) states ". . . groundwaters frequently contain high total counts of bacteria with no coliforms. Such waters pass Interim Drinking Water Regulations but technical judgment must conclude these are not acceptable as potable waters." Wisconsin health standards similarly related only to the presence of the coliform indicator. While non-coliform colonial growth may or may not reflect pathogenicity, it does suggest that the well water has been recently exposed to the surface soil or atmospheric environments. The cause of such growth may be as innocuous as incidental organic or construction contamination; but it could also reflect a broken seal or too shallow a well of only marginal safety, which may be a pathway for contamination from surface runoff. The admission of ubiquitous soil organisms, not coliform in nature, to a drinking water supply may not violate present health codes, but it certainly should be reason for concern since some of these organisms have been shown to be opportunistic pathogens. Such results, while relatively frequent, are certainly not the norm for good quality drinking water.

Similarly, concentrations of nitrites and nitrates and/or chlorides in a residential well considerably higher than those of one's neighbors may not violate present water quality standards, but should induce at least further investigation.

The number of samples tested so far does not demonstrate a correlation between the presence of coliform colonies, nitrates, and/or increased concentrations of chlorides. This is not surprising since these components have different mobility characteristics in soils and need not necessarily be derived from the same source. For example, one lake community studied revealed extremely variable chloride data with well samples varying from a chloride concentration of less than 1 to more than 100 mg/l. There was no spatial pattern to these results that could equate the data to groundwater movement. There also was no correlation with nitrates which might have implied septic system sources of the chlorides. However, nitrates would be low if the drainfield was in the water table and nitrification inhibited by low oxygen. Road salting is not reported to be a local practice and apparently few homeowners have water softeners.

In the communities studied, many people had no idea how deep their wells were; when they were installed;

whether they met current code specifications; or when they had been tested for safety. Of those households where this information was available, there was no evident relationship between the depths of wells and instances of suspect samples.

Motivation of the voluntary participants in the program may influence the nature of the test results obtained. There is no specific evidence available to suggest just what prompts an individual householder to participate in or to avoid the program (but it is evident that very few of the homeowners encountered ever have their water tested after their well is installed). Some people signed up because they perceived the chance to take advantage of a bargain in the free screening test. Others may have done so from a sense of responsibility for theirs and their friends' health. But others have stated that they avoided the program because they suspect a water quality problem and don't want it identified if remedial expenses may be involved.

CONCLUSIONS

Data analysis from this small, non-random sample indicates a surprising number of well water samples containing coliforms, indeterminate bacterial populations, and/or unusually high nitrate-nitrite or chloride concentrations. Of 351 screening tests in these communities from 21 to 94 percent of the samples tested were of suspicious water quality. The overall average was 57 percent. Even if the abnormal and unexplained chloride data for community "F" is deleted from the data, 52 percent of the 351 samples remain suspicious. While further investigation is indicated, particularly with respect to extended incubation times and the identification of atypical bacteria observed, it is evident that better residential water supply management is needed in lake communities. This improved management should begin with efforts to encourage homeowners to have their well water tested on an annual or semiannual basis, and to avail themselves of professional remedial services where indicated.

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PHOSPHORUS INACTIVATION: A SUMMARY OF KNOWLEDGE AND RESEARCH NEEDS

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ABSTRACT

Phosphorus (P) precipitation and inactivation are lake improvement techniques which can lower the lake P content sufficiently to retard algal growth. P precipitation has been shown to have at least short-term effectiveness in improving lake trophic state. Longer term lake improvement is more likely to occur through control of P release from lake sediments (inactivation); in one case, significant improvement over a 5-year period occurred. A basis for determining a safe maximum dose to lake sediments has been developed. There is a need to improve application procedures to lower treatment costs. Toxicity problems seem to be absent, but additional research is needed. Long-term monitoring of P inactivation treatments to establish cost-effectiveness should be encouraged.

INTRODUCTION

Phosphorus (P) precipitation/inactivation is a lake improvement technique to lower the P concentration in the water to a level sufficient to reduce standing crop and/or productivity of planktonic algae. This is accomplished by either removing P from the water column (precipitation) or by controlling P release from nutrient-rich sediments (inactivation). This treatment is used to accelerate lake improvement after nutrient diversion, particularly in those cases where internal P release represents a significant contribution to the P budget (Cooke, et al. 1977; Larsen, et al. 1979). While this procedure, as it is now understood, may be effectively used to remove material (e.g., phosphorus, silt) from the water column, its principal objective is long-term control of P release from lake sediments through the sorptive action of a layer of colloidal aluminum hydroxide on the sediments.

Our knowledge of this lake improvement technique has been summarized by Cooke and Kennedy (1980a, b), and the reader is referred to these works for details of effectiveness, dose determination, application procedures, problems with toxicity, case histories, and costs. Funk and Gibbons (1979) have also reviewed the technique, including a useful discussion of costs.

The purposes of this paper are to briefly describe what we know of this technique and what we need to know.

PHOSPHORUS PRECIPITATION-INACTIVATION

Phosphorus Precipitation

It is now well-known that the addition of aluminum salts to lake waters, principally aluminum sulfate ($Al_2(SO_4)_3$), will bring about a prompt lowering of phosphorus concentration. Removal of P can occur as $AlPO_4$ precipitate, by sorption to the surface of $Al(OH)_3$ polymers or floc (which is formed when $Al_2(SO_4)_3$ is added to water with carbonate alkalinity), or by entrapment of particulate P in the $Al(OH)_3$ floc. Removal of particulate and inorganic P is dependent upon the quantity of floc and upon pH (Eisenreich, et al. 1977; Cooke and Kennedy, 1980a). Removal of dissolved organic molecules which contain P is considerably less effective (Browman, et al. 1973, 1977; Eisenreich, et al. 1977), a factor which could be of major significance in the prompt return of blue-green algal blooms since some nuisance species of this phylum synthesize alkaline phosphatases at low inorganic P levels and thereby remove P from dissolved organic molecules (Heath and Cooke, 1977).

At this writing, we are aware of 28 lake and pond treatments to remove (precipitate) or inactivate P, 19 of which have had the objective of P removal (see Table 1 of Cooke and Kennedy, 1980a, which lists all of these treatments and summarizes their results). Of these 19, only four appear to have any amount of published information (Jernelov, 1970; Peterson, et al. 1973; May, 1974; Funk and Gibbons, 1979). In each of these cases it was clearly demonstrated that addition of aluminum sulfate (ferric alum and aluminum sulfate in the case of May, 1974) can effectively remove a large percentage of P in the water column and bring about at least short-term improvement in lake trophic state. Documentation of any long-term lake improvement, with the exception of Horseshoe Lake, Wis. (Peterson, et al. 1973) is not now available for any of these P precipitation or removal projects. At Horseshoe Lake, a dose of 2.1 g Al/m^3, as slurried aluminum sulfate, was applied in May 1970. Figure 1 illustrates the reduction in hypolimnetic P concentration which was achieved, and the control of P through the summer of 1972. According to Born (1979), P has never reached the levels found before treatment, although it has increased slightly each year since application.

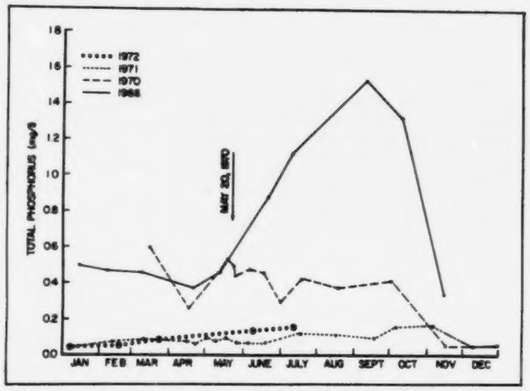

Figure 1. — (Cooke & Kennedy)

Control of Phosphorus Release from lake Sediments

More recent treatments of lakes with aluminum salts have been based on the recognition that lake sediments can be an important source of P to the water column (May, 1974; Kennedy, 1978; Cooke, et al. 1978; Gasperino and Soltero, 1978; Knauer (this book); Dominie (this book)), and that long-term lake improvement may occur if this important P source is also given long-term control. That is, aluminum salts are added primarily to cover lake sediments with a P-sorbing floc of $Al(OH)_3$, and not for P removal from the water column. A stated but as yet untested hypothesis of this approach is that longer, more complete control of P release will occur in proportion to the amount of aluminum added.

Only two lakes, Dollar and West Twin in Ohio (Cooke, et al. 1978; Kennedy, 1978; Cooke, 1979), have received sufficient monitoring to substantiate the conclusion that a large dose of aluminum sulfate to the lake sediments, well in excess of that needed to remove P from the water column, will bring about a long-term improvement in lake trophic state. In July 1974, the hypolimnion of Dollar Lake (A = 2.2 ha., Z = 3.9m was treated with 9 metric tons of liquid aluminum sulfate. One ton was added to the surface. In July 1975, West Twin's (A = 34 ha., Z = 4.4m) hypolimnion was treated with 100 metric tons of liquid aluminum sulfate (26 g Al/m^3). In both cases a procedure for adding a maximum safe dose, described in Cooke, et al. (1978), Kennedy (1978), Cooke and Kennedy (1980a), and Kennedy and Cooke (this book) was followed. Figures 2 and 3 illustrate the results.

P content was sharply lowered and has remained so for West Twin through 1980. Dollar Lake, a seepage lake, had a slight increase in P content in 1978, probably reflecting accumulation of P from cultural sources. Table 1 lists the changes in trophic state (using the Carlson, 1977, Trophic State Index), and shows that the lakes are now in the mesotrophic range. Significant lake improvement thus occurred, as evidenced by higher transparency, lower total P concentration, and decreased planktonic algae. Dollar and West Twin have remained in this improved state for 6 and 5 years (through summer, 1980), respectively. East Twin also improved since it obtains most of its water from West Twin. All lakes have more macrophytes than before, perhaps due to the higher transparency.

At this writing, our knowledge about the long-term effectiveness of both P precipitation and P inactivation in controlling nuisance planktonic algae is not complete enough to warrant extended conclusions about longevity of effect, cost-effectiveness, or any long-term detrimental changes. Phosphorus removal seems to be effective for at least 2 years (Horseshoe Lake), and P inactivation seems to bring about significant lake improvement for at least 5 years.

Table 1. Mean (May-September) Carlson Trophic State Index (from surface measures; adapted from Cooke, 1979, based on Total Phosphorus

Year	West Twin	East Twin*	Dollar
1971	57.58	53.68	no data
1972	62.75	58.91	no data
1973	61.36	56.48	64.31
1974	59.84	58.89	no data
1975	55.85	57.14	50.22
1976	52.36	56.62	50.65
1978	44.25	47.27	47.79
1980**	46.81	46.81	46.81

Table 1 (continued). Mean (May-September) Carlson (1977) Trophic State Index (Cooke, 1979), based on Secchi Disk transparency.

Year	West Twin	East Twin*	Dollar
1968	ND	ND	66.3
1969	50.0	51.6	ND
1971	61.0	50.4	ND
1972	48.3	52.8	ND
1973	43.2	49.0	63.8
1974	49.9	50.5	ND
1975	51.4	51.9	50.7
1976	46.7	51.4	47.9
1978	46.4	45.5	47.8
1980**	43.5	45.0	48.2

*Untreated downstream reference lake
**Based on average of 2 measurements, July 1980

At the Dollar-West Twin (Ohio) hypolimnetic applications, the Wisconsin system was modified to include on-shore aluminum sulfate storage, and a distribution pipeline from shore to a mid-lake platform where application barges would return to re-fill. Another adaptation of the Wisconsin system was that of Dominie (this book), who used a three-compartment tank truck mounted on a barge, to add a mixture of aluminum sulfate and sodium aluminate to a soft-water lake. May (1974; this book) added ferric alum to ponds by suspending blocks of chemical in the water and allowing them to dissolve.

Actual application to the lake is usually accomplished by dividing the lake into small, well-marked sections of known area and volume. Pre-application calculations then permit the barge operator to know the volume of chemical to be added to each section.

Thus, adequate application procedures have been developed for the P removal-P inactivation lake improvement method. However, these procedures (Cooke and Kennedy, 1980a) are expensive (labor costs range from 1 to 4 man-days per hectare for the six treatments for which such data are available) and tedious, and represent an obstacle to the general use of this method. As well, equipment design and construction are often difficult. Some new and effective procedures need to be developed.

Determination of Effective Dose of Aluminum

Many lake treatments with aluminum sulfate apparently had no basis at all for the dose added. Regrettably, much of this work is thus of little value in attempting to understand the results and in developing a systematic procedure.

Most of the projects in which P removal was the primary objective had dose based upon Al/P ratios, following the model from water treatment plants. The amount of aluminum needed to remove the desired amount of P was determined in jar tests and the total aluminum dose was then obtained by multiplying amount of aluminum needed by the P content of the lake. This procedure produced low doses of aluminum (0.5 to about 10 g Al/m^3), and usually adequate P removal.

The first stated attempt to control P release from sediments was the Cline's Pond Project in April 1971 (Sanville, et al. 1976), in which 10 g Al/m^3 as sodium aluminate, plus HCl to prevent high pH, were added to the pond surface. The treatment was successful for at least 1 year in controlling P concentration, but algal blooms then returned. While the authors suggest that their short-lived success may have been caused by continued external loading, sediment disturbance, and breakup of the floc, an equally plausible explanation is simply insufficient dose.

What is a sufficient dose to control P release from lake sediments for a prolonged period? The answer is not known but it is assumed that it is necessary to put as much aluminum hydroxide over the sediments as possible, short of causing adverse environmental impact. Kennedy and Cooke (1974) and later Kennedy (1978) were the first to suggest a basis for determining a maximum dose for P inactivation. A maximum dose of aluminum sulfate was defined as that amount which

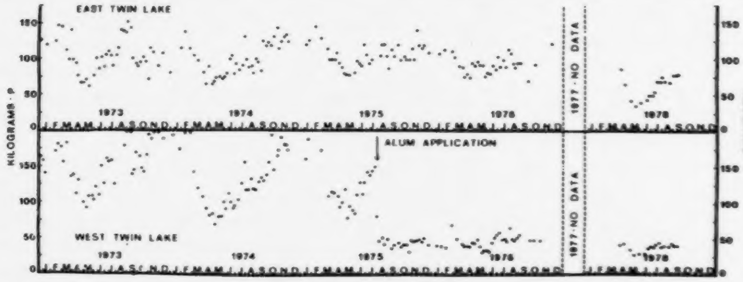

Figure 2. — (Cooke & Kennedy)

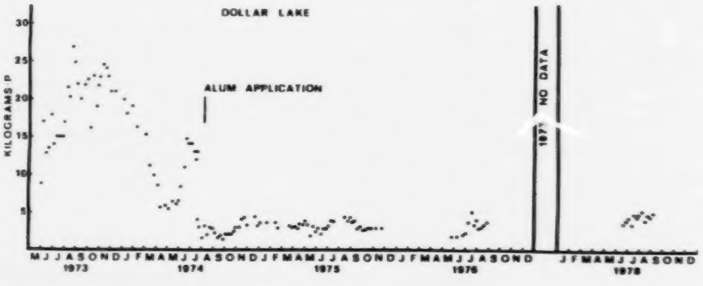

Figure 3. — (Cooke & Kennedy)

Or, the maximum dose is that amount which can be added until pH 6.0 is reached, a pH at which little dissolved aluminum appears. Details of determining dose are completely described for an actual lake treatment in Cooke, et al. (1978), for any lake in Cooke and Kennedy (1980a), and in additional detail by Kennedy and Cooke (in this book). For softwater lakes, the dose would be very small using this definition. In such cases, as exemplified by the work of Dominie (in this book), a mixture of sodium aluminate and aluminum sulfate is added and pH and dissolved aluminum remain well within acceptable limits.

Thus dose procedures for P precipitation or removal and P inactivation are known, and are sufficiently well understood to allow any user of the technique to apply the proper amount of chemical.

Application Methods

The basic application system, which has been used throughout the 12-year history of this method, was designed for the 1970 Horseshoe Lake treatment (Peterson, 1973). Aluminum sulfate was stored on board a barge as a solid, mixed with water and then pumped to a manifold trailing behind the barge, and applied to either surface or deep waters (later projects) as a slurry. The usual procedure has been to treat surface waters for P removal, and to add aluminum to hypolimnetic waters where P inactivation is the object. One reason for a hypolimnetic application is that there will be no exposure of epilimnetic and littoral biota to the chemical. Also, less volume equals less cost.

Toxicity

Unlike herbicides, adding aluminum salts to lake waters does not involve a substance whose toxicity to plants is the factor in controlling algae. The procedure works by lowering P concentration to a level which controls productivity. However, adding aluminum to lakes can pose a hazardous condition to the biota if lake pH is sufficiently lowered (addition of aluminum sulfate) or raised (addition of sodium aluminate) to bring about solubilization of aluminum hydroxide and an increase in dissolved aluminum. The toxicity of aluminum has been reviewed by Burrows (1977) who, along with Everhart and Freeman (1973), pointed out that few investigations of aluminum toxicity have considered the complex chemistry of aluminum in water. The amount of dissolved aluminum which will appear in the water after a treatement is pH dependent and will vary from lake to lake as a function of lake alkalinity and amount of the aluminum salt which has been added. It is in this dissolved form that aluminum could be hazardous. Cooke and Kennedy (1980a, b) and Kennedy and Cooke (1974; this book) have thus suggested that a maximum dose of aluminum sulfate be one in which pH does not fall below 6.0 nor dissolved aluminum increase above 50 ug Al/m^3.

No toxicity to fish has been observed at any of the full-scale lake treatments. However, no systematic investigation has been made of this possibility, or of the possibility of aluminum in fish muscle tissue. The hazards posed here are small because of the very low toxicity of aluminum to humans (Berry, et al. 1974). Narf (1978) has investigated benthic invertebrate populations following the several lake treatments in Wisconsin and reports no adverse changes. Moffett (1979) found a significant persistent (at least 3 years after a hypolimnetic application) reduction in the H′ species diversity of planktonic microcrustacea after the West Twin treatment. Causes and importance of this observation are unknown.

Despite indications of little or no hazards after adding aluminum salts to lakewater, few systematic investigations of aluminum toxicity to aquatic populations or communities are available. It may be of particular importance to note that acid precipitation may bring about sufficient lowering of a treated lake's pH so that previously insoluble aluminum hydroxide yields significant quantities of dissolved aluminum. This could be of real significance in soft water lakes.

RESEARCH NEEDS

The evidence strongly supports the belief that an aluminum application for control of P release can be an effective, and apparently long-lasting, method of controlling algae. Procedures for determining dose and applying the chemical are available. The toxicity of aluminum to aquatic communities has not been evaluated with the exception of Narf's work on benthos and Moffett's study of planktonic microcrustacea. This is a critical need since the technique is effective in controlling algae, and it may be used with increasing frequency and with maximum doses. It is important to note here that our research need is not with toxicity to laboratory organisms, but with the short- and long-term impacts on the actual level of biological organization to which aluminum salts are applied, namely the community level. This means we need studies of changes in community metabolism, mineral cycling, species diversity, and other attributes of lake state. The use of the LD or the Maximum Acceptable Toxic Concentration (MATC) will not be very useful for evaluating a lake improvement agent since it is the lake which is treated, not animal species.

Rapid application techniques might make the P inactivation method less costly. The use of suspended blocks of alum, as described by May (1974; this book) is one approach which could prove to be economical and effective. As well, for somewhat larger systems, use of shore-based high velocity hoses could reduce manpower costs. For larger lakes, barge applications now seem to be most cost-effective. There is a great need to develop new, innovative methods of aluminum application.

Is P inactivation cost-effective? We do not know the answer to this question partly because we lack published information about it. Also, support for long-term monitoring of this and other lake restoration techniques is very small. Accurate cost-benefit cannot be accurately assessed without such monitoring.

SUMMARY

The effectiveness and longevity of P inactivation following nutrient diversion has been demonstrated on a few lakes, but the majority of demonstrations of control of P release are new and most data are as yet unreported. This symposium will add greatly to our knowledge. It is clear at this point that if sufficient aluminum is added, control of P release will occur and remain effective at least 5 years, and that this approach, rather than P precipitation, is the method of choice for most situations.

A basis for determining dose for the P precipitation and the P inactivation techniques has been developed which gives adequate assurance that at least immediately toxic levels of aluminum will not be reached. Aluminum application may pose a long-term hazard to lake biota, but to date there has been little systematic research about this possibility. At present, based on a long-term monitoring of benthic invertebrates and short-term monitoring of planktonic microcrustacea, the data are equivocal about change in lake communities.

Costs and benefits of the technique need further assessment. Funk and Gibbons (1979) and Cooke and Kennedy (1980a) indicate a wide range of costs, but without long-term monitoring, costs vs. effectiveness cannot be stated.

It is recommended that future research include these projects:

1. Monitoring of change in lake trophic state after P inactivation and P precipitation treatments;
2. Studies of changes in field and experimental (enclosures, microcosms) communities after an aluminum application;
3. Development of rapid aluminum application techniques; and
4. Evaluation of the possible effects of acid rainfall upon the pH and subsequent release of dissolved aluminum in treated lakes.

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CONTROL OF TOXIC BLUE-GREEN ALGAE IN FARM DAMS

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ABSTRACT

Light, warmth, and polluted water in a lake or inland water reservoir are likely to lead to Cyanophyte algal blooms, which often include toxic strains. Preventing or reducing the entry of soluble plant nutrients and carbohydrates is a necessary preliminary to the restoration of the water quality. This includes controlling the quality of the drainage from both point and nonpoint sources. Inactivation of the nutrients already present in the system is a second essential step. In small farm dams this inactivation has been achieved by pre-summer dosing of the water with ferric alum blocks suspended in the water. This treatment has proved to be a functional and ecologically satisfactory method of reducing or controlling toxic algae in these cases. Similarly, for larger water storages, liquid alum has proved effective, perhaps in conjunction with sodium aluminate, so that aluminium hydroxide floc is produced near the sediment. Destratification or hypolimnion aeration is another treatment producing good results.

CONDITIONS PROMOTING GROWTH OF TOXIC BLUE-GREEN ALGAE

Slow moving or stagnant water frequently contains soluble plant nutrients and carbohydrates. If it also is subject to adequate light and warmth, then heavy growths of both bacteria and algae are likely to develop.

The bacteria use the carbohydrates and oxygen, increasing CO_2 levels and decreasing O_2 content of the water. Anaerobic conditions (Sylvester and Anderson, 1964) free solutes and nutrients from the previously enriched sediments. This includes a rise in the concentration of phosphorus (Patrick and Khalid, 1974).

Algal growth becomes stimulated by an increase in available soluble nutrients, including phosphorus, obtained either from this bacterial action or from any local direct drainage. The algae then proceed to increase photosynthetic carbon uptake, resulting in an increased pH value (King, 1970.)

These conditions usually occur in deep waters of stratified dams, but may occur also during drought in small dams, which then suffer severe evaporation and consequent increase in mineral content, together with lowered aeration.

Conditions for algal growth then may include low light intensity (because of the depth of the water or its turbidity), low O_2 levels, and high CO_2 levels, which, because of high pH values, may be of low availability.

It has been shown that each of these conditions (Holm-Hansen, 1967; Stewart and Pearson, 1970; King, 1970, respectively) favors the growth of Cyanophytes as contrasted with green algae. Hence massive growth of Cyanophytes might well be expected.

Further, many species of blue-green algae develop specialized gas vacuoles, which seem to allow vertical mobility, so that the plant can benefit from both deeper water (i.e., higher nutrient levels) and shallower water (i.e., more light). Hence, these species are likely to be among those which develop in excessive numbers.

A further property of certain blue-green algae is their ability to fix atmospheric nitrogen, and this again aids their growth under certain conditions.

Thus it seems that, starting with adequate light, warmth, and enriched water, together with the naturally good distribution mechanism of freshwater organisms, it is almost inevitable that dams will be subject to blooms of blue-green algae.

These blooms are perhaps unsightly, but our main concern is that toxic strains occur among the most prevalent bloom-forming species. In Australia these species are *Anacystis cyanea* (Kuetz.) Dr. & Dail. (*Microcystis aeruginosa* Kuetz.), *Anabaina circinalis* Rabenh. (including *A. flos-aquae* as far as concerns the literature of toxic Cyanophyte algae, see May 1980), and *Nostoc spumigena* (Mert.) Drouet (*Nodularia spumigena* Mert.). These are known to kill horses, cows, sheep, fowls, turkeys, laboratory guinea pigs and mice, and probably various wild animals including birds and fish. The lethal agent is an endotoxin which affects the liver and can cause death, in some cases within a few minutes.

Each of these bloom species may occur in the field either in almost pure culture, or, in the case of *Anacystis* and *Anabaina*, at times as codominants.

The particular species which develops in a dam may depend on the preceding crop of algae growing there. Thus Lam and Silvester (1979) record that *Microcystis (Anacystis)* inhibits the growth of *Anabaina* sp; they suggest this might be caused by the production of inhibitory extracellular products by *Microcystis*. Fitzgerald (1964) stresses the transient effect of products such as these, since there is a very rapid development of succeeding species of algae.

Figure 1 shows the periods of occurrence of *Anacystis* and of *Anabaina* in Carcoar Dam, N.S.W. Each year *Anacystis* was present before, and continued after, the *Anabaina*. From these figures it seems that in this case the conditions favoring the growth of *Anabaina* are perhaps more limited than are those allowing the growth of *Anacystis*. This variation could link with nutritional requirements; Fitzgerald (1969) showed that in co-dominant bloom of *Anabaina* sp. and *Microcystis* sp. (*Anacystis* sp.) phosphorus was a limiting factor for the growth of one genus while nitrogen was for the other.

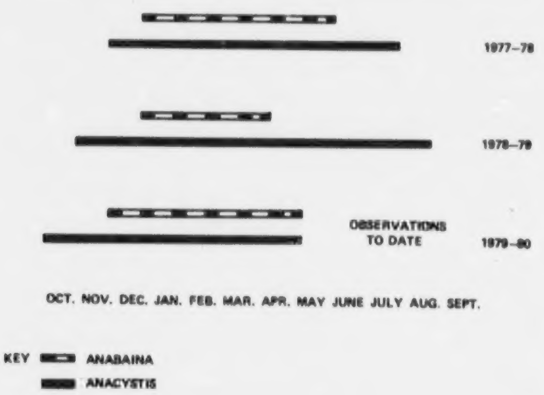

Figure 1. — Total occurrences of *Anacystis cyanea* and *Anabaina circinalis* in all collections from Carcoar Dam, New South Wales, from October 1977 to May 1980.

CONTROL OF TOXIC BLUE-GREEN ALGAE

Biological control would be the most satisfactory of all treatments. Some work has indicated blooms could be controlled by particular viruses (Safferman and Morris, 1964). but this treatment has not so far been practical.

Many different algicides have been used in the past to rid water of unwanted blue-green blooms, 1 ppm of copper sulfate possibly being the best and cheapest. However, a bloom treated by an algicide can recur in as short a time as a week, and an algicide may cause much ecological harm either where used or downstream. Hence it would be preferable to use a method of decreasing or preventing the occurrence of such blooms, especially if this method were ecologically acceptable.

To break sequence of development of a bloom stimulated by the described conditions, it follows that one would need to limit one or more of the following: (1) light, (2) temperature, (3) carbohydrates, (4) high pH, (5) low oxygen levels, or (6) high levels of nutrients.

1. Light could be excluded by covering the water with lightproof material, but this is often impractical, particularly for large areas. Plastic covers (Anon. 1979) or even numerous floating black ping-pong balls have been used to cover small areas.

2. It is quite impractical to reduce temperature in the open, especially in large storage reservoirs.

3. Carbohydrates are so prevalent, provided by either land or aquatic plants or animals, macrophytes, or plankton, that it is only for excessive quantities such as from sewerage or urban drainage that control is at all practical. Here, of course, it is highly desirable in order to restrict later intense de-oxygenation caused by bacterial action.

4. High pH. Since any heavy photosynthetic growth will increase pH, and since most blue-green algae control seems directed to increasing the alternative growth of green algae, control of this factor for long periods seems likely to be impractical.

This leaves two areas — "low oxygen levels" and "high levels of soluble nutrients" — as possible avenues to decrease blooms of Cyanophytes. Both avenues have been tried.

5. Overcoming low oxygen levels. To overcome low oxygen levels, a compressor is often used to release air (or oxygen) into the hypolimnion water, fairly close to the reservoir bottom. The resultant artificial mixing or destratification of a water body has been very widely employed. This treatment not only reduces the amount of nutrient solution coming from the sediment, but also breaks the thermal stratification of the storage water, if present, and makes the all-over water temperature more uniform. Improvement in the quality of the bottom water often overcomes additional problems, such as high concentrations of iron and manganese, and perhaps of hydrogen sulfide, which affect later water use. Further, low oxygen concentrations in the water released from low outlets in a dam wall could also damage biological communities downstream. Destratification in Australia is relatively new (Bowles, et al. 1979).

An early report on the advantages of destratification by aeration is given by Howard (1972). He reported the widespread effect within a reservoir of aeration applied at only one site. Tolland (1977) advises that, while destratification is useful in overcoming existing water quality problems, it is better used in a preventive rather than curative role.

A submerged hypolimnion aerator which preserved thermal stratification was described by Fast, et al. (1975). This was designed to aerate the deep water but did not introduce the rich inorganic nutrients of the hypolimnion into the photic zone. It also had the added advantage of maintaining a suitable habitat for cold-water fish. Certain problems for fish, because of nitrogen gas supersaturation, need to be guarded against with this treatment, particularly with deeper lakes.

The high costs of installing the apparatus make these treatments unsuitable for small dams.

6. Prevention of high levels of soluble nutrients. The first and obvious way to achieve nutrient prevention has been to stop adding nutrients by way of drainage, sewerage discharge, aerial spraying or verge-pollution, i.e., to control all sources of added nutrients, of both point and nonpoint origin.

When sewerage and/or polluted drainage is diverted from a water storage, eutrophication in the latter decreases markedly. This treatment, unfortunately, is expensive and the benefit may be delayed if the sediment is already highly polluted, so that it continues to release nutrients to the water during periods of anaeroby for some time after the diversion.

Purifying the discharge itself (usually sewage) is also expensive, but usually not nearly as difficult as diverting it. This also leads to obvious benefits.

Catchment areas should be freed as much as possible from nearby manurial material, and distant siting arranged for contaminating activities such as abattoirs and piggeries. Leaving the verge of a water storage clear of animal use (such as by using water troughs rather than dam edges for stock drinking) is a simple expedient, but worthwhile.

It is perhaps interesting to diverge here to an observation about a river, the Peel River, N.S.W., which I am at present studying with serial collections. I have a number of observation stations along the length of the river, on which a dam has recently been constructed. During the time that the dam wall has been under construction, and before the flow of the river was affected at all, there was massive disturbance of soil at the construction site. It was just below this that *Anabaina circinalis* was recorded, not frequently, but more often than at higher or lower stations. Doubtless this was caused by increased nutrients from the soil, even though conditions in the flowing river were not conducive to heavy growth.

Controlling blooms by a constant natural removal of nutrients from the water system by biota is obviously an extremely satisfactory method. Constant removal of fish containing the nutrient minerals, might seem feasible. Seasonal blooms, however, do not provide a year-round basis for a food chain. Perhaps alternative food could be made available at other times of year.

An interesting study by Weir (1976) investigated the possibility of developing macrophyte beds of the angiosperms *Typha* and *Eleocharis* by planting zones of them around a bloom-susceptible northern N.S.W. lake and its connected swamp. These plants would remove mineral nutrients, stabilize the mud banks, and reduce erosion of soil. It was further suggested that they might also be grown in artificial floating rhizome beds, and could be managed by cropping routinely to maximize the uptake and removal of nutrient. Hopefully, this would also lead to some profit, as the cropped weed could be used for stock feed, since it was shown that periodic cropping reduced fibre content and maintained unusually high levels of phosphorus and sulfur in the plant material. Thus, digestibility of the crop was increased while removal of nitrogen and phosphorus from the water continued at a high level.

A similar scheme is in use in Holland where bulrush reed ponds allow treatment of sewage from holiday camps with a periodic input of sewage at weekends; the reeds are cropped annually and later burned to return nutrients to the agricultural system (de Jong 1975, cited Weir, 1976). This sort of scheme is likely to be particularly effective in controlling the entry of nonpoint sources of nutrients from the edges of a reservoir. It does not suffer from the disadvantage of using a floating plant such as water hyacinth, where pest quantities of growth can affect navigation.

Since extremely nutrient-enriched sediment may prolong enrichment to the water, sometimes it has been considered necessary either to dredge (Hudson and Marson, 1970), de-silt (as is usual in farm dam management), or cover it, as suggested by Theis, et al. 1978, who placed fly ash on the eutrophic sediment. Another reported cover for the sediment was a nylon-covered fabric supported on a polyurethane grid (Anon. 1972).

Next, attention has focused on which of the soluble nutrients should be particularly reduced or inactivated to control the growth of unwanted Cyanophytes.

Trace elements could be considered, but with any normal catchment area it would be extremely difficult to control contamination by such small quantities as needed. Nicholas (1980), however, has suggested that sodium tungstate may inhibit the growth of *Anabaina* since this material is antagonistic to molybdenum, a trace element necessary for growth of the alga.

Most work has suggested that phosphorus or nitrogen limitation is likely to be the most functional approach to controlling unwanted blooms. Of these elements, controlling nitrogen, even where it may be a limiting factor, seems impractical since some of the toxic blue-green algae (e.g. *Anabaina)* are nitrogen-fixing. It is useless to spend one's efforts reducing the nitrogen level if the algae are going to replace this element from atmospheric nitrogen.

Most studies seem to concur that controlling the level of phosphorus in the water is a practical method of reducing Cyanophyte blooms, and that reducing phosphorus at wastewater treatment plants is a necessary, practical, and economic plan. This is despite it being known that surplus ("luxury") phosphorus can be held by the plant and that sometimes heavy growths are recorded when the phosphorus concentration in the water is low.

My work was directed toward controlling phosphorus in small farm dams, particularly during summer, since this is where and when we in Australia suffer most stock losses.

My first work on this project was on a small inland dam at Braidwood, N.S.W. Here it appeared that blooms (of *Anacystis* and /or *Anabaina*) occurred whenever the level of phosphorus rose to 0.5 ppm or higher (May, 1972). As a control measure I hoped to use a chemical which, applied before the summer rise in phosphorus levels, would combine with this phosphorus before the unwanted algae could absorb it. I applied alum and block ferric alum (Alumina ferric R.), at a combined concentration of 200 mg/1. This treatment is effective through the absorption of phosphorus by the aluminum hydroxide floc, which is formed when these treatment chemicals are added to alkaline water (May and Baker, 1978). The blocks were much easier to handle than is loose alum. They were suspended in the water from floats, to prevent them sinking into the underlying mud, and were replaced at intervals. Following this treatment, for the first time in 5 years no blooms developed in the treated dam (May, 1974) although apparently some did occur in untreated dams in the same district and in the same year. Later, other dams with similar histories were treated similarly and again no blooms developed (May, 1974).

Prior to the next field experiment, preliminary *in vitro* investigations were carried out by Harvey Baker, my co-author in some of this work (May and Baker, 1978). This study indicated that, at the alkalinities usually present in dams, treatment with aluminum sulfate or ferric alum blocks reduced a range of initial phosphorus levels to below 0.5 mg/1, a concentration deemed critical.

For the field study a series of farm dams, all of which had recently suffered from toxic algae, was studied concurrently, Some dams were untreated and were considered as controls, while others had a single pre-summer treatment with ferric alum blocks, and the third group of dams was treated this way at recurrent intervals (May and Baker, 1978).

All of these dams showed a summer rise in the concentration of total phosphorus, but the presence of the alum reduced the phosphorus levels and also decreased the incidence of algal blooms. The dosage used (50 mg/1) was evidently sufficient to reduce but not eliminate bloom occurrence; a somewhat higher dose is therefore now recommended (100 mg/1 — this equals approximately 9 μg/l aluminum). It appears that the earlier (Braidwood) dosage (200 mg/1) would be unnecessarily high.

The alum treatment not only reduced excessive growth of these bloom algae, it also increased the total number of algal species. In addition, the average number of species per collection occurring in the treated dams increased, i.e., there was greater algal diversity. Evidently this alum treatment leads to conditions more like those prevailing in pre-eutrophic times.

This treatment is proving satisfactory in reducing bloom formation in our numerous small dams, many of which previously presented a repeated threat to our stock. It is most satisfying to feel that this treatment, in contrast to the use of algicides, is ecologically pleasing. Indeed, this method of nutrient inactivation appears to offer a promising routine for farm dam protection and restoration.

Other speakers at our Symposium are to tell of the dosing of larger volumes of water with alum, with or without sodium aluminate, and the generally successful effect this has had in reducing the level of phosphorus and the incidence of Cyanophyte blooms. It should be noted that besides reducing the concentration of phosphorus already in the water, or later freed from the sediment, aluminum sulfate can also reduce the pH of the water. If this drops below — and stays below — 6, this condition alone makes the growth of toxic algae less likely.

Any treatment, by alum or aeration, can have its benefits masked by enriched incoming drainage. It follows that these treatments are more effective where such external sources of enrichment are absent or minimal; their use in other cases is probably limited to making the end result less damaging than it would have been otherwise.

Hence, for good control one needs both lake treatment and also drainage management, both localized (point) and general (nonpoint).

In conclusion, lake restoration depends on a series of processes:

1. a. Diverting all massive polluted water drainage (such as sewage) from the storage area, or at least cleaning this water of its pollutants, both mineral and carbohydrate.

b. Reducing nonpoint (runoff) drainage enrichment as far as practical. This source of pollution is of varying relative importance in different waterways. Any residual drainage of this sort is probably best cleansed by using some sort of littoral harvesting.

2. The best method so far available to control the annual enrichment of the water internally from an already polluted sediment depends on the size of the impoundment.

a. If the water impoundment is small, treatment with suspended block alum at a dosage of 100 mg/1 seems best. This also is likely to cope with some general runoff pollution.

b. If the water impoundment is large, either alum treatment or destratification or hypolimnion aeration should be suitable.

3. If these treatments prove inadequate, i.e., in dams with excessively polluted water and sediment, then direct treatment of sediment, such as by dredging or covering the sediment, may also be necessary.

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ALUMINUM SULFATE DOSE DETERMINATION AND APPLICATION TECHNIQUES

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ABSTRACT

Nutrient diversion alone does not always adequately reduce in-lake phosphorus concentration because of nutrient-rich sediments. Certain lakes and reservoirs may continue to experience nuisance algal blooms and will require additional restorative steps. The phosphorus precipitation/inactivation technique is a procedure to remove phosphorus from the water column and to control its release from sediments. The salts of aluminum have long been used in advanced wastewater treatment to remove phosphorus and this technology was logically extended to lake rehabilitation. However, specific guidelines for dose calculation and application to lakes and reservoirs are lacking. An objective of all aluminum treatments, although often unstated, is to control phosphorus release from bottom sediments. The suggested approach to dose determination allows maximum application of aluminum to bottom sediments and thus emphasizes long-term control of phosphorus recycling. Such a dose can be calculated directly from the alkalinity of the water to be treated. Titration of several lake-water samples of varying alkalinity will allow the establishment of the relationship between residual dissolved aluminum, alkalinity and dose which can then be employed for lake-scale applications of alum to lakes and reservoirs. Application equipment and procedures will depend on site characteristics and treatment objectives. General equipment requirements include lakeside storage, a distribution pipe, and an application barge and manifold. In addition to phosphorus removal and control, alum may be used to meet other restoration objectives including the treatment of problem inflows and the reduction of particulate concentrations.

INTRODUCTION

Aluminum sulfate application for postdiversion phosphorus control in eutrophic lakes is an increasingly popular management tool. A logical adaption of water and waste treatment technology, aluminum addition provides a direct ameliorative methodology for high phosphorus concentrations in lakes and small reservoirs. However, the current popularity of the method, attributable to its simplicity and the fact that it produces immediate reductions in lake phosphorus concentrations, has misled many lake managers to view alum as a panacea. Despite 12 years and 25 reported uses (Cooke and Kennedy, 1980), aluminum treatments remain more of an art than a well understood technological alternative in lake restoration.

Problems arise from the failure of many lake managers to establish limnologically appropriate objectives or to fully understand the aqueous chemistry of aluminum. Although the importance of phosphorus recycling from anaerobic sediments in delaying the response of lakes to reduced external phosphorus inputs (e.g. Cooke, et al. 1978, Larsen, et al. 1976) has prompted the use of aluminum sulfate, only recently has the control of sediment phosphorus been specifically identified as the primary treatment objective. The use of aluminum to precipitate phosphorus from the water column, still often identified as the primary objective of many lake treatments, provides little more than short-term relief. Sound decisions concerning the relative importance of recycling from anaerobic sediments must be made prior to treatment, and treatment methodologies must concentrate on sediment phosphorus control if long-term effectiveness is to be realized.

Confusion concerning dose determination methods for lake treatments is a related problem. Three approaches, each dictated by treatment objective, have been followed to date. The first involves incremental additions of aluminum to aliquots of lake water until a predetermined phosphorus removal efficiency is attained (e.g. Peterson, et al. 1973). This dose, which is then volumetrically scaled for lake application, clearly optimizes treatment for phosphorus removal from the water column. A second similar method, also optimizing dose for phosphorus removal, is modeled after dose determination procedures employed in waste treatment facilities. With pH controlled, aluminum additions are made to constantly mixed lake water samples until optimum phosphorus removal is achieved. The A1/P molar ratio at maxium phosphorus removal and the phosphorus concentration of the lake to be treated are then used to determine lake-scale doses (e.g., Peterson et al. 1974). These controlled laboratory conditions are quite different than those which occur during lake treatment, however, and can often underestimate effective lake doses (Kennedy, unpublished). The third method, initially employed by Kennedy (1978) and later by Cooke, et al. (1978), maximizes aluminum input to sediments, as dictated by the buffering capacity of the overlying water, and thus emphasizes long-term phosphorus control as a primary treatment objective.

If it can be demonstrated that nutrient-rich anaerobic sediments will be a significant source of phosphorus long after external sources are reduced, then aluminum treatments must be targeted against phosphorus exchanges at the sediment-water interface. The purpose of this paper is to provide dose determination guidelines for such treatments and to suggest appropriate application procedures and possible management strategies employing aluminum salts.

ALUMINUM CHEMISTRY

A considerable body of information concerning the chemistry of aluminum is available. Since it is not the purpose here to provide a comprehensive review of this information, the reader should consult such reviews as that of Hayden and Rubin (1974) for more detailed discussions. However, a basic understanding of the aqueous chemistry of aluminum, which is essentially the chemistry of aluminum hydroxide, is necessary for making dose determination decisions.

The addition of aluminum salts (e.g. aluminum sulfate) to water, which initially results in the hydration of aluminum ions, is followed by a series of hydrolysis reactions resulting in a decreased pH, and ultimately, in the formation of low solubility aluminum hydroxide precipitate. In natural waters a secondary consequence is a decrase in carbonate alkalinity. Aluminum hydroxide is amphoteric and thus is converted to the soluble aluminate ion in basic solution.

Significant for dose determination is the fact that the distribution of aluminum species is pH dependent (Figure 1). While insoluble aluminum hydroxide predominates between pH 6 and 8, soluble species occur at higher (Al(OH^4) and lower (Al(OH)$_2^+$ then Al^{+3}) pH. As aluminum is added to alkaline lake water, hydrogen ion concentration increases, alkalinity is titrated and pH decreases (e.g. Figure 2). Initially, at low aluminum dose, pH changes are small. If solution pH remains alkaline, the dissolved aluminum concentration (i.e. Al(OH^4) will be predictably high. Further aluminum additions decrease pH, favoring the formation of insoluble aluminum hydroxide precipitate and dissolved aluminum concentrations decrease. As aluminum additions continue, dissolved aluminum concentrations again increase in acidic solution, with Al^{+3} predominating below pH 4.

The importance of pH change is thus of direct concern in dose determination since, in addition to obvious consequences for exposed biota, the pH of treated lake waters will dictate the concentration of potentially hazardous soluble aluminum species, and the quality and quantity of aluminum hydroxide polymer. Although the toxicity of aluminum for aquatic biota is poorly defined (Burrows, 1977), some conservative estimates are available. Concentrations of dissolved aluminum below 52 μgAl/l had no obvious effect on rainbow trout (Freeman and Everhart, 1971) or salmon (Peterson, et al. 1974, 1976). These findings prompted Kennedy (1978) and Cooke, et al. (1978) to adopt 50 μgAl/l as a safe upper limit for posttreatment dissolved aluminum concentrations. Dose was defined as the maximum amount of aluminum which would still ensure low (μgAl/l) concentrations. Since, based on solubility, dissolved aluminum concentrations, regardless of dose, would remain below 50 μgAl/l in the range pH 5.5 to 9.0, a dose producing a posttreatment pH in this range could also be considered environmentally safe.

The formation of large aluminum hydroxide polymers or floc, essential for the deposition of added aluminum, would also be promoted in this pH range. Rapid removal of floc from the water column is of concern since prolonged suspension of fine aluminum hydroxide particulates further complicates the question of toxicity and treatments targeted against specific areas (e.g. anaerobic sediments) would be adversely impacted by the mixing and dispersion of floc.

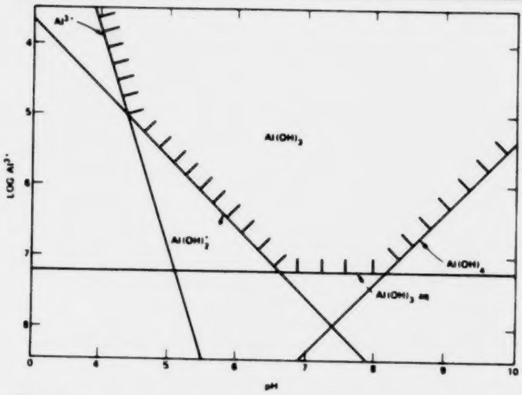

Figure 1. — pH — dependent distribution of aluminum species (Eisenrich, et al. 1977).

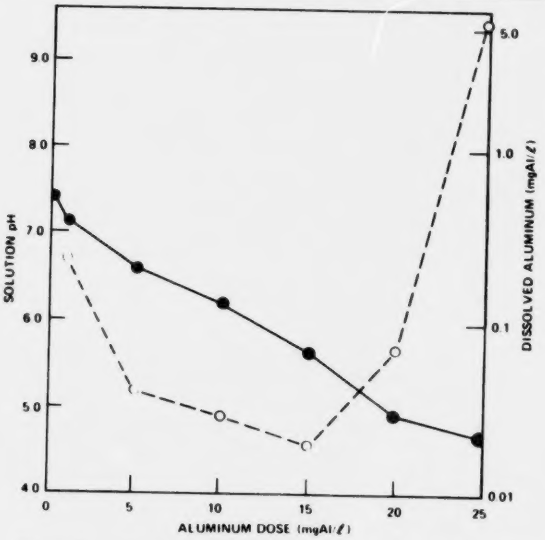

Figure 2. — Changes in pH (closed circles) and post treatment dissolved aluminum concentration (open circles) following additions of aluminum sulfate to lake water (initial total alkalinity of 98 mg $CaCO_3/\lambda$; pH 7.3).

PHOSPHORUS REMOVAL

The long-term effectiveness of alum treatments will depend on the ability of deposited aluminum hydroxide to retain phosphorus at the sediment/water interface and thus curtail its internal recycling. Secondary, short-term benefits can be realized if water column phosphorus concentrations can be reduced during treatment. Phosphorus removal can occur by coagulation/entrapment of phosphorus-containing particulates, precipitation of $AlPO_4$ (Recht and Ghassemi, 1970) or by sorption of phosphorus on the surfaces of aluminum hydroxide polymers (Eisenreich, et al. 1977). Successful removal of particulates will depend on the quality of floc produced, which in turn is related to pH and aluminum dose. Precipitation and sorption, both influenced by pH and phosphorus concentration (Stumm and Morgan, 1970), are apparently related processes since sorption appears to occur by the formation of aluminum-ion-phosphate bonds at the surface of aluminum hydroxide polymers (Hsu, 1965).

At high phosphorus concentrations, such as those encountered in wastewater facilities and low pH, $AlPO^4 4$ is the predominant reaction product. However, eutrophic lakes are characteristically alkaline and, despite having biologically high phosphorus concentrations, are relatively low in phosphorus. At low phosphorus concentrations and higher pH, OH^- reacts more readily with aluminum than does phosphate and thus aluminum hydroxide is the expected product. Therefore, phosphorus removal from the water column will be primarily by entrapment and sorption. Failure to obtain maximal phosphorus removal at the stoichiometric Al/P molar ratio of 1.0 supports this suggestion. This is particularly true in the case of lake treatments. For example, maximum phosphorus removal from Cline's Pond water occurred at Al/P molar ratios ranging from 5.7 to 7.2 (Peterson, et al. 1976). Al/P molar ratios in excess of 525 were required to achieve 90 percent P removal from unfiltered Lake Mendota epilimnetic water (Eisenreich, et al. 1977).

Physical factors influencing phosphate sorption include floc size and settling rate. As floc size increases, specific surface area decreases. Increased floc size also increases settling rates and thus decreases contact time between the floc and the surrounding lake water. Therefore, phosphorus removal will be highest on the immediate area of aluminum addition since pH decreases would be greatest here and floc size would be small. As floc size increases during mixing and settling, phosphorus removal efficiency decreases.

Aluminum hydroxide gels deposited on anaerobic lake sediments would be exposed to high interstital phosphorus concentrations and relatively low pH. Phosphorus removal would continue by further sorption/precipitation and the Al/P molar ratio of deposited gels would decrease with prolonged exposure to high phosphorus concentrations. Kennedy (1978) eluted laboratory-produced gel with phosphorus solutions at pH 6, 7, and 8 (Figure 3). Phosphorus removal was pH dependent, and although initially high, was minimal at Al/P molar ratios ranging from 2 to 4. Therefore, effectiveness and longevity will depend on the amount of aluminum deposited relative to sediment phosphorus concentration and the rate at which phosphorus is made available.

Few direct field evaluations of the effectiveness of deposited aluminum hydroxide gels in retaining phosphorus have been reported. An exception is Dollar Lake, Ohio, which was treated hypolimnetically in 1974 with 10 tons (197 mg Al/l) of aluminum sulfate (Kennedy, 1978). Phosphorus concentrations of water samples collected immediately above capped treated and untreated anaerobic sediments were compared during summers 1974, 1975, and 1976, with differences expressed as percent reduction (Figure 4). Percent reduction averaged about 90 percent following treatment but decreased to about 75 percent and 65 percent in 1975 and 1976, respectively.

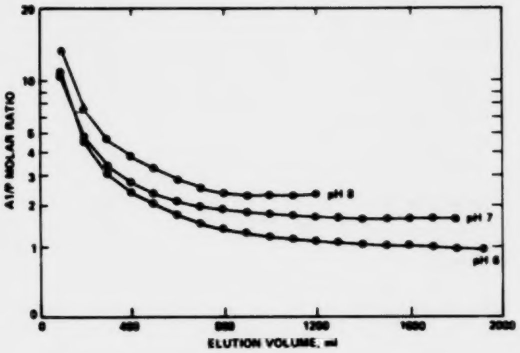

Figure 3. — Changes in the Al/P molar ratio of aluminum hydroxide gels eluted with phosphate buffer at pH 6, 7, and 8 (Kennedy, 1978).

DOSE DETERMINATION

Lake managers have employed a number of different dose determination methodologies based on two major objectives: Phosphorus removal from the water column and control of phosphorus release from sediments. Immediate reductions in phosphorus concentration, while often desirable, are generally secondary to long-term treatment objectives. However, adoption of proper dose determination methodology will allow calculation of a dose accomplishing both objectives.

Phosphorus precipitation/sorption, aluminum hydroxide formation, and dissolved aluminum concentration are pH dependent. In the range pH 6-8, dissolved aluminum concentrations will be minimal while phosphorus removal and floc formation will be maximal. Although the control of sediment release requires maximal aluminum deposition, maximum additions to the lake will be dictated by conditions in the water column and by changes resulting from treatment. Since excessive additions of aluminum will produce undesirable side effects (e.g. low pH and alkalinity, and high dissolved aluminum concentrations), an optimum dose would be that dose which reduces pH to about 6.0. This optimum dose would maximize the amount of aluminum deposited over sediments as dictated by lake conditions.

Data collected from two Ohio lakes treated hypolimnetically with doses of aluminum sulfate which

Table 1

	Dose		Percent Phosphorus Removal			
Lake	mg Al/l	TP	SRP	SUP	PP	Reference
			"Maximum" Dose			
Dollar Lake	7.5(T)	83(T)	96(T)	62(T)	65(T)	Kennedy, 1978
	19.7(H)	91(H)	97(H)	65(H)	63(H)	
West Twin Lake (Hypo. Treatment)	22.6	92(H)	99(H)	--	--	Cooke, et al., 1978
			Other Dose Methodologies			
Mytajarvi	12.2	40	92	--	--	Dunst, et al. 1974
Langsjon	4.5	57	92	--	--	Jernelove, 1970
Cline's Pond	10.0	-0	-0	--	--	Sanville, et al. 1976
Lake of 4 seasons	1.8*	--	90	--	--	Dunst, et al. 1974
Horseshoe Lake	1.8	25(E)	--	--	--	Petersen, et al. 1973
		14(H)	--	--	--	
Pickerel Lake	7.0	-25-30	--	--	--	D.R. Knauer, per. comm.
Liberty Lake	0.4	--	96	--	--	Funk, 1977
State Rearing Pond	11.0*	90	--	--	--	Dunst, et al. 1974
Medical Lake	--	55(E)	80(E)	--	--	A. Gasperino, per. comm.
Lake Mendota**	16.2	79(E)	90(E)	17(E)	--	Eisenreich, et al. 1977
		81(H)	72(H)	30(H)	--	
Lower Nashotah**	16.2	43(E)	100(E)	36(E)	--	Eisenreich, et al. 1977
		40(H)	100(H)	29(H)	--	Eisenreich, et al. 1977
Lake Wingra**	16.2	42(E)	8(E)	1(E)	--	Eisenreich, et al. 1977
Little John**	16.2	27(E)	82(E)	1(E)	--	Eisenreich, et al. 1977

Note: T = whole lake, E = epilimnion, H = hypolimnion.
* mg Al/m^2.
** Results of laboratory experiments using lake water.

were calculated based on similar considerations, indicated that optimum doses also substantially reduced water column phosphorus concentrations (Table 1). While the removal of soluble reactive phosphorus (SRP) was similar to that obtained using other dose determination methods (e.g. those employing Al/P molar ratios and additions for maximum phosphorus removal), the use of a maximum (Kennedy, 1978) dose markedly increased the removal of total (TP) and soluble unreactive (SUP) phosphorus. Therefore, optimum doses will effectively accomplish both treatment objectives and thus allow for the establishment of a single dose determination method.

The following simple method for determining optimum doses for aluminum sulfate applications requires limited laboratory equipment and expertise, and can be easily implemented by local lake managers.

Procedure:

1. Obtain representative water samples from the lake to be treated. Care should be exercised in selecting sampling stations and depths since significant heterogeneities, both vertical and horizontal, commonly occur in lakes. Samples should be collected as close to the anticipated treatment date as possible.
2. Determine the total alkalinity and pH of each sample. Total alkalinity, an approximate measure of the buffering capacity of lake water, will dictate the amount of aluminum sulfate (or aluminum) required to achieve pH 6 and thus optimum dose. Additional chemical analyses can be performed, depending on the specific needs of the investigator. For example, phosphorus analyses before and after laboratory treatment would allow estimation of anticipated phosphorus removal effectiveness.
3. Determine the optimum dose for each sample. Initial estimates of this dose, based on pH and alkalinity, can be obtained from Figure 5. More accurate estimates should be made by titrating samples with fresh stock solutions of aluminum sulfate of known aluminum concentration using a standard burette or graduated pipette. The concentration of stock aluminum solutions should be such that pH 6 can be reached with additions of 5 to 10 milliliters per liter of sample. Samples must be mixed (about 2 minutes) using an overhead stirring motor and pH changes monitored continuously using a pH meter. Optimum dose for each sample will be the amount of aluminum, which when added, produces a stable pH of 6.0.
4. The relationship between total alkalinity and optimum dose can be determined using information from each of the above titrations by plotting optimum dose as a function of alkalinity. This relationship will allow determination of dose at any alkalinity within the range tested.

APPLICATION TECHNIQUES

Estimation of the relationship between optimum dose and alkalinity provides a means by which laboratory determined doses may be scaled for lake

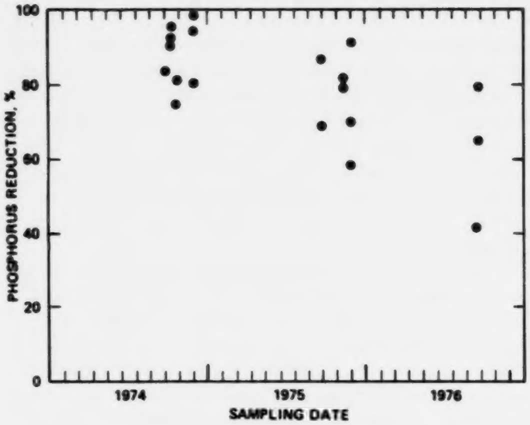

Figure 4. — Percent reduction in total phosphorus concentration of sediment seepage water collected above aluminum treated Dollar Lake sediments (Kennedy, 1978). Dollar Lake was treated hypolimnetically with aluminum sulfate July 1974.

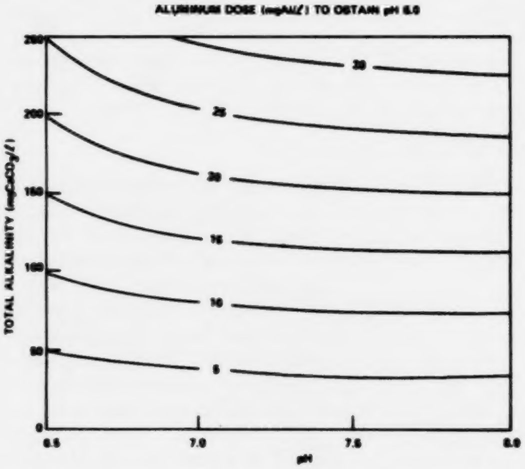

Figure 5 — Estimated aluminum sulfate dose (mg Al/λ) required to obtain pH 6 in treated water of varying initial alkalinity and pH. Based on equations in Ferguson and King (1977) and assuming insignificant phosphorus concentrations.

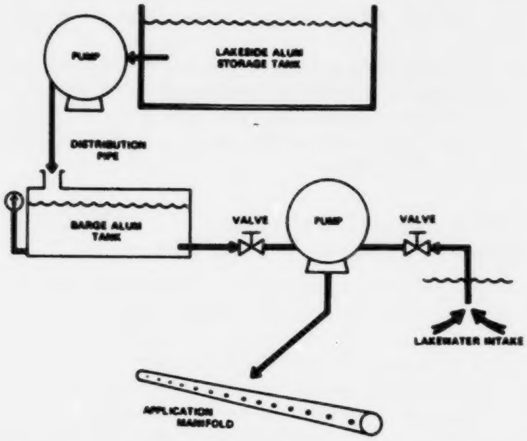

Figure 6. — Basic components of a lake application system (Cooke and Kennedy, 1980).

treatment. Total treatment dose can be determined by calculating a volume-weighted mean total alkalinity for segments of the lake (e.g. depth strata) having similar pH. Careful consideration should be given to the extent to which mixing can be accomplished since this will determine the volume of water to be treated. For example, surface treatments of deep lakes would, depending on application method and equipment, involve only waters in upper depth strata.

Once the total application dose is determined, provisions must be made for proper distribution with respect to depth and volume. This is most easily done by establishing a treatment grid system and calculating dose allocations for each treatment quadrant (Kennedy, 1978; Cooke, et al. 1978; Funk, 1977; Peterson, 1973). The grid system will also facilitate field procedures if the intersections of grid lines are marked with coded or numbered buoys.

Aluminum sulfate may be purchased in granular form or as a liquid and lake managers must determine which is more convenient. The use of liquid alum simplifies field procedures since it does not require mixing and may be pumped directly from tank trucks to application equipment. It does, however, involve further dose calculations to account for liquor density and temperature (see Cooke and Kennedy, 1980, for a discussion of these calculations). Liquid alum has the disadvantage of not being readily available in many areas of the country, as well as presenting storage problems. Granular alum, while more readily available and easier to store, must be dissolved prior to use, thus complicating field procedures.

Application equipment systems employed to date have consisted of a shorebased storage/mixing facility, a distribution pipe, an application barge, and an application manifold (Figure 6). The exact design of an application system, while generally requiring these components, will reflect the lake manager's specific objectives and site characteristics. In general, treatment will involve pumping alum from a storage or mixing tank through a floating pipe, tube or hose to a smaller storage tank on an application barge and then to a distribution or application manifold. Provisions for pumping both alum and lake water to the manifold will allow for flash mixing before discharge and provide a means for adjusting alum discharge rate. Mixing should be accomplished by movement of the barge and/or by turbulence behind the manifold. Additional mixing during surface treatments could be attained by positioning the manifold behind the propeller of the barge motor. In any case, mixing should be maximized.

Applications may be made to surface water or at predetermined depth(s) depending on treatment approach. Surface treatments, while less complicated and time consuming, will be less effective in reducing hypolimnetic phosphorus concentrations, a concern in cases requiring phosphorus removal. Surface treatment will also expose near-surface organisms to reduced pH. Applications at depth (e.g. immediately above the hypolimnion) may be made by suspending the manifold below the application barge on a rigid frame.

Treatment costs are highly variable and will depend on availability of chemicals and equipment, chemical costs, available labor, lake size, and dose. Reported costs for several lake treatments are reviewed in Cooke and Kennedy (1980).

LAKE MANAGEMENT STRATEGIES USING ALUMINUM SULFATE

High phosphorus and low dissolved oxygen concentrations do not necessarily indicate that aluminum sulfate application will have a beneficial impact following nutrient diversion. Lakes which have not had a long history of excessive nutrient and organic loads may respond immediately to curtailed loadings (Schindler and Lee, 1974), while lakes receiving substantial inputs of clay in addition to nutrients may contain sediments with high sorptive capacities for phosphorus. Extensive growths of macrophytes may, through senescence and decay, continue to be the dominant phosphorus source long after nutrient diversion (Barko and Smart, 1979). Therefore, the relative importance of sediments as a phosphorus source must be assessed prior to attempting costly aluminum sulfate applications.

Nutrient budget calculations provide a simple means for assessing the importance of internal loading. Cooke, et al. (1977) employed such calculations to data from two Ohio lakes, one of which was subsequently treated with aluminum sulfate, and determined that internal sources accounted for 65 to 100 percent of the summer increase in phosphorus content. While such calculations do not specifically identify sources within the lake, they can, when viewed in light of other observations, indicate the possible need for in-lake phosphorus control measures. For example, the presence of thick organic, nutrient-rich sediments and a low macrophyle density in lakes experiencing significant internal phosphorus recycling would at least suggest that aluminum sulfate application would hasten lake recovery following reductions in external nutrient loads.

How effective and long lasting are aluminum treatments for sediment-phosphorus control? This will depend on sediment characteristics and the quantity of aluminum added and, at best, can only crudely be estimated. Since phosphorus sorption, which decreases as Al/P molar ratio decreases, would effectively cease at an Al/P molar ratio of 2 to 4, 1 mg of aluminum could potentially remove 0.6 to 0.3 mg of phosphorus. However, a number of other factors, including disruption of the floc layer (Browman, et al. 1977), deposition of sediments subsequent to treatment, and phosphorus uptake by floc during settling would affect this estimate.

If there is reason to believe that optimum dose applications would supply insufficient amounts of aluminum to sediments (e.g. low alkalinity lakes), combined aluminum sulfate/sodium aluminate treatments could be considered (Wirth, et al. unpubl.). Doses for such treatments could be considered (Wirth, et al. unpubl.). Doses for such treatments can be estimated by modifying the laboratory procedure. Sodium aluminate increases pH and additions must be balanced by additions of aluminum sulfate. Once optimum pH is reached using aluminum sulfate only, further additions would be possible by adding 3 moles of aluminum as sodium aluminate for every mole of aluminum added as aluminum sulfate. Doses calculated in this manner would also allow larger additions to sediments with the greatest potential for phosphorus release, such as those deposited near nutrient-rich inflows.

Aluminum sulfate treatments may also be employed for purposes other than sediment-phosphorus control. Ree (1963) treated three California water supply reservoirs and tributaries to reduce turbidity caused by storm runoff from construction sites and thus reduce particulate loads to water treatment filter beds. Alum may also be used to coagulate and sediment organic particulates, including algae (Lin, et al. 1971), as a means of reducing oxygen demand following intense algal blooms or macrophyte die-off. Applications to littoral areas, while having little inhibitory effect on the growth of macrophytes (Dunst, et al. unpubl.), could retain phosphorus released during decay (Funk, et al. 1977) and thus reduce phosphorus inputs to pelagic areas following herbicide treatments. Periodic inputs of phosphorus and organic particulates could be reduced by temporarily retaining and treating storm runoff in urban areas (Shapiro and Pfannkuch, 1973).

CONCLUSION

Phosphorus control by in-lake chemical treatment is only one of many lake restoration techniques available to lake managers; the decision to use aluminum sulfate must be based on careful evaluation of lake conditions and management objectives. Although such treatments provide a simple method for removing particulates and phosphorus from lake water, they are more appropriate in management situations requiring control of phosphorus release from eutrophic sediments. The dose determination method described here allows calculation of doses which maximize aluminum input to sediments. The degree to which such treatments provide long-term control over internal phosphorus recycling is not adequately documented; future restoration efforts should include provisions for long-term evaluation.

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A COMPARISON OF TWO ALUM TREATED LAKES IN WISCONSIN

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ABSTRACT

Two alum treated lakes, Pickerel and Mirror, are compared. The polymictic nature of Pickerel Lake rather than the alum appeared to be responsible for observed changes in phytoplankton changes in the dimictic Mirror Lake. Pickerel Lake demonstrated only minor fluctuations in total-P concentrations following treatment. Pre- and post-treatment comparison of phytoplankton biomass indicated a reduction following the alum application. However, after holomixis occurred in July, phytoplankton biomass was similar to or greater than the pre-treatment year. Mirror Lake total-P concentrations were reduced from 90 μg/l to 20 μg/l . The phytoplankton biomass during the spring and fall periods has decreased and the principal nuisance alga *Oscillatoria agardhii* has been eliminated.

INTRODUCTION

The positive results of aluminum applications for lake restoration in Sweden (Jernelov, 1970) encouraged similar treatments in the United States. The use of aluminum for removing phosphorus from eutrophic lake waters is an extension of water and waste treatment processes. Aluminum hydroxide has a high capacity for removing dissolved and suspended phosphorus materials under conditions that are common to lakes. In addition, the use of aluminum salts is relatively inexpensive and has a low toxicity to most forms of aquatic life.

The objective of the alum treatment at Pickerel and Mirror Lakes was to rapidly remove available phosphorus from the lake and at the same time prevent release of phosphorus from the lake sediments, thereby limiting the growth of planktonic plants. The decision to treat Pickerel Lake was based on a history of previous algal problems and associated fish winter kills. Mirror Lake was treated to enhance the recovery rate following a nutrient diversion project.

SITE DESCRIPTION

Both Mirror and Pickerel Lakes are glacial seepage lakes approximately 19 kilometers apart situated in outwash plains formed during the recession of the Cary ice sheet of the Pleistocene Glaciation about 10,000 to 14,000 years ago. The physical characteristics of each lake are presented in Table 1. The volume of both lakes is similar although Pickerel Lake has four times the surface area of Mirror Lake. The mean depth of Mirror Lake, however, is three times that of Pickerel Lake.

An important morphological difference between Mirror and Pickerel Lakes is their respective relative depths. The relative depth (Z_r) is defined as the maximum depth as a percentage of the mean diameter (Hutchinson, 1957). The larger the Z_r value, the more stable the lake. Pickerel Lake has a Z_r of 0.5 percent and is considered polymictic while Mirror Lake has a Z_r of 2.3 percent and prior to artificial mixing in the fall of 1977, was considered meromictic. Since the fall of 1977, Mirror Lake has been artificially circulated for several weeks each spring and late fall.

In 1972, the influx of total-P to Pickerel Lake from surface runoff, ground water and direct precipitation was calculated to be 24 kg which is equivalent to a phosphorus loading rate (P) of 0.13 gm/m^2.yr (Hennings, 1978). The allochthonous sources of phosphorus to the lake are diffuse and the direct drainage basin is forested with only one permanent dwelling.

Mirror Lake is located in the city of Waupaca and received an annual allochthonous total-P influx of 15 to 20 kg in 1972 and 1973 (Knauer, 1975; Peterson, 1974). In 1976 a storm sewer diversion project reduced the allochthonous sources of phosphorus by 50 to 60 percent. As a result of the diversion project, the P has been reduced to 0.12 gm/m^2 yr, very similar to Pickerel Lake.

RESULTS

Phosphorus — Pickerel Lake

Liquid aluminum sulfate was applied in Pickerel Lake on April 17, 1973. The application rate was 170 kg Al/ha to yield a concentration of 7.3 mg Al/l in the lake. The liquid alum was released near the surface and at mid-depth to ensure a reasonable distribution in the water column.

As a result of the aluminum sulfate additon, the pH declined from 8.2 to 7.1 and total alkalinity was reduced by approximately 25 mg/l (110 to 85 mg/l as $CaCO_3$). There was no immediate effect on the phosphorus concentrations in the lake (Figure 1a). Total-P (weighted mean) increased from 28 µg/l in early June 1973 to 73 µg/l by late July 1973.

Following holomixis the total-P concentration in Pickerel Lake was reduced to 10 µg/l The reduction of total-P during fall overturn has been noted in other alum treated lakes, e.g., Horseshoe Lake, Wis. and Medical Lake, Wash. (Peterson, et al. 1973; Gasperino, et al. 1980). The phosphorus dynamics appear to be different in Pickerel Lake, however, when compared to the other alum treated lakes. In Horseshoe, Medical, and Mirror Lakes, the alum floc in the sediments prevented sediment phosphorus from being released into the overlying waters. In Pickerel Lake, the weighted mean total-P/^{2}concentrations increased during the fall months from 10 g/l in September 1973 to 30 µg/l by November 1973 (Figure 1a) and 50 µg/l by March 28, 1974. The increasing total-P concentrations during the winter months suggest the alum floc had not completely prevented sediment phosphorus release. An analysis of aluminum concentrations in sediment cores from Pickerel Lake before and after treatment indicated the floc had been redistributed towards the center of the lake basin following a series of holomictic occurrences throughout the summer and fall of 1973. A large area of lake sediments was subsequently interacting with the overlying waters without the benefit of an aluminum floc covering.

Phosphorus — Mirror Lake

In the year following the storm sewer diversion, the annual average phosphorus concentraion, 90 µg/l , showed little change from previous years, 88 and 93 µg/l in 1972 and 1973, respectively (Smith, et al. 1975). During the summer, total-P concentrations in the epilimnion were typically 20 µg/l; however, in the hypolimnion total concentrations increased to 550 µg/l. The hypolimnetic total-P concentrations appear to be the result of sediment-phosphorus release when the bottom waters were anaerobic (Mortimer, 1941; Kamp-Nielsen, 1974). Data from nutrient regeneration chambers placed on the lake sediments indicated a total-P release rate of 1.8 mg P/m^2 day during 1977 (Knauer and Garrison, 1979). However, that rate becomes limited as a concentration of 550 µg/l is approached and a sediment-water phosphorus equilibrium is achieved.

Alum was applied to Mirror Lake on May 17, 1978, since it appeared that sediments could potentially provide a significant source of phosphorus. Previous studies by Ryding and Forsberg (1977) and Welch (1977) suggest that as a result of internal phosphorus loading, lakes from which large allochthonous nutrient sources were eliminated recovered very slowly. The alum was applied at a rate of 337 kg Al/ha, and at a depth of 3 meters to achieve an aluminum concentration of 6.6 mg/l.

As a result of the alum treatment, weighted mean total-P concentrations were reduced from 90 µg/l to 20 µg/l , a 78 percent reduction (Figure 1b). Owing to an algal bloom during the treatment date, much of the total-P was in the particulate fraction and the dissolved reactive-P (DRP) concentration was less than 4 µg/l The phosphorus reduction following the alum treatment was the result of physical entrapment of particulate-P (algae). Carbon, nitrogen, and phosphorus data from sedimentation traps at the 12 m depth confirmed the fact that algae were carried to the sediments with the alum floc.

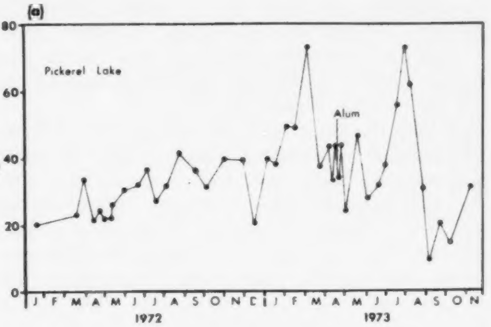

Figure 1a. — Weighted average total phosphorous for Pickerel Lake during 1972 and 1973. The lake was treated with aluminum sulfate on April 17, 1973.

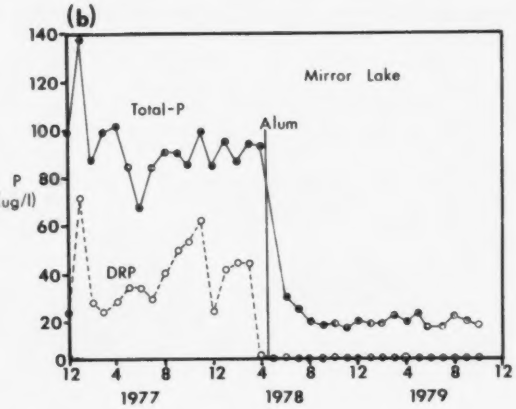

Figure 1b. — Weighted average total and dissolved reactive phosphorus for Mirror Lake during 1977, 1978, and 1979. The hypolimnion of the lake was treated with aluminum sulfate on May 17, 1978.

As indicated in Figure 1b, the alum treatment has been successful in preventing release from the sediments. Weighted mean total-P concentrations have remained at 20 µg/l and DRP has been undetectable for at least 2 years following the alum treatment.

Phytoplankton — Pickerel Lake

In 1972, the algal biomass was relatively constant, 6 to 7 mm^3/l (Figure 2a), throughout the summer and fall. During the summer months, the major components of the algal biomass were from the phyla Chlorophyta and Pyrrhophyta. After holomixis in mid-September, a short pulse of *Microcystis aeruginosa* was observed, followed by a diatom pulse that dominated the algal biomass through October. During the month of November, the dominance shifted from diatoms (*Stephanodiscus*) to the green alga (*Ankistrodesmus fractus* (Figure 3).

Following the alum treatment in mid-April 1973, the algal biovolume remained approximately 3 mm^3/l through July (Figure 2b). This was half the 1972 biovolume concentrations; the alum treatment apparently was effective. However, in late July the lake completely mixed and by the end of August the biomass reached 43 mm^3/l. The algal assemblage was dominated by *Microcystis aeruginosa* from August through mid-October (FIgure 3).

In 1972 and 1973, *M. aeruginosa* was not observed in the surface waters until the lake mixed. It is possible that *M. aeruginosa* may have been present at or near the sediment surface. Light measurements taken in Pickerel Lake during the summer of 1973 with a submarine photocell indicated 1 percent of surface light was present at the sediment surface through July. Positive primary productivity measurements were also recorded at the 4½ m depth during 1973. Our data also showed that immediately after holomixis in 1972 and 1973, the biovolume of *M. aeruginosa* was similar, 4.5 mm^3/l. Our data suggest that as a result of holomixis, *M. aeruginosa* was distributed throughout the water column in both years. Following holomixis in mid-September, 1972, the environmental conditions (temperature, light, etc.) were suboptimal for *M. aeruginosa* and the population never expanded. Fallon and Brock (1980) have also reported a rapid decline of *Microcystis* during late September in Lake Mendota.

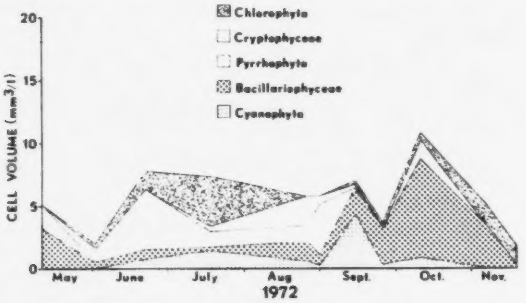

Figure 2a. — Algal biomass in Pickerel Lake during 1972 at 0.5 meters.

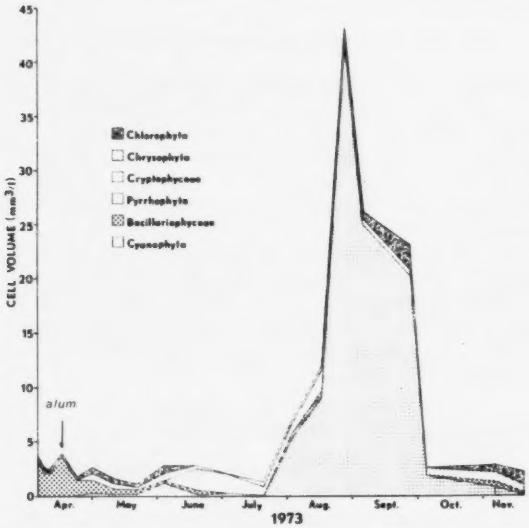

Figure 2b. — Algal biomass in Pickeral Lake during 1973 at 0.5 meters. The lake was treated with aluminum sulfate on April 17, 1973.

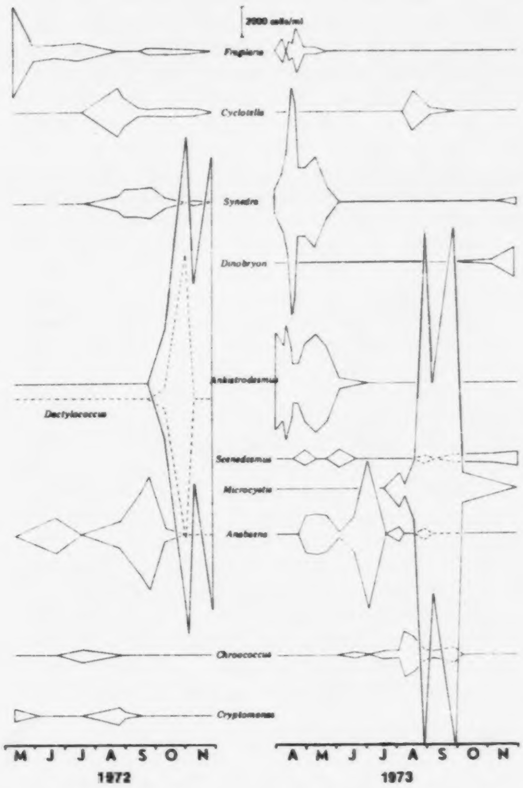

Figure 3. — The seasonal succession of major phytoplankton genera, 1972 and 1973 for Pickerel Lake at 0.5 meters.

In 1973, holmixis occurred in late July and *M. aeruginosa* was distributed throughout the water column when environmental conditions were more favorable for growth. The biovolume increased from 4.5 mm^3/l following holomixis to 43 mm^3/l by late August. As in the previous year, the population rapidly declined in late September.

Phytoplankton — Mirror Lake

Mirror Lake did not experience the summer algal problems that are typical in many eutrophic lakes. The problem alga in Mirror Lake was *Oscillatoria agardhii* (Figure 4). This species dominated the phytoplankton assemblage during the late fall and early winter months and at spring overturn (Figures 5a and 5b). Although this alga was present during the summer, it remained only in the lower metalimnion, albeit in large concentrations. The occurrence of *O. agardhii* throughout the lake during the fall overturn was owing to the redistribution of the metalimnetic population and not an increase in the growth rate.

The biovolume of *O. agardhii* was similar in the spring of 1977 and 1978, 6 mm^3/l (Figures 5a and 5b). Following the hypolimnetic alum treatment on May 17, 1978, a decline in the *O. agardhii* biovolume was observed during the fall of 1978 and 1979 and subsequent spring of 1979 (Figure 5c). *O. agardhii* was not present during the spring of 1980 nor was it present in metalimnion during the summer of 1980.

At times, *O. agardhii* has dominated the metabolism of Mirror Lake. Other studies (Smith, et al. 1975) have shown that during the summer, BOD's in the metalimnion of Mirror Lake were five times higher than elsewhere in the water column as a result of the *O.agardhii* population. The reduction of phytoplankton biomass and the elimination of *O. agardhii* as a result of the alum application have substantially improved the water quality of of Mirror Lake.

In summary, the introduction of liquid aluminum sulfate into the water column of Pickerel Lake for the purpose of lowering the phosphorus concentration and reducing algal biomass was not successful. It is our opinion that alternative techniques for alum addition to polymictic lakes, e.g. plowing into the sediments, should be researched. The application of alum to the water column of dimictic lakes appears to be a successful technique to improve water quality, providing the phosphorus loading to the lake has been reduced to an acceptable level.

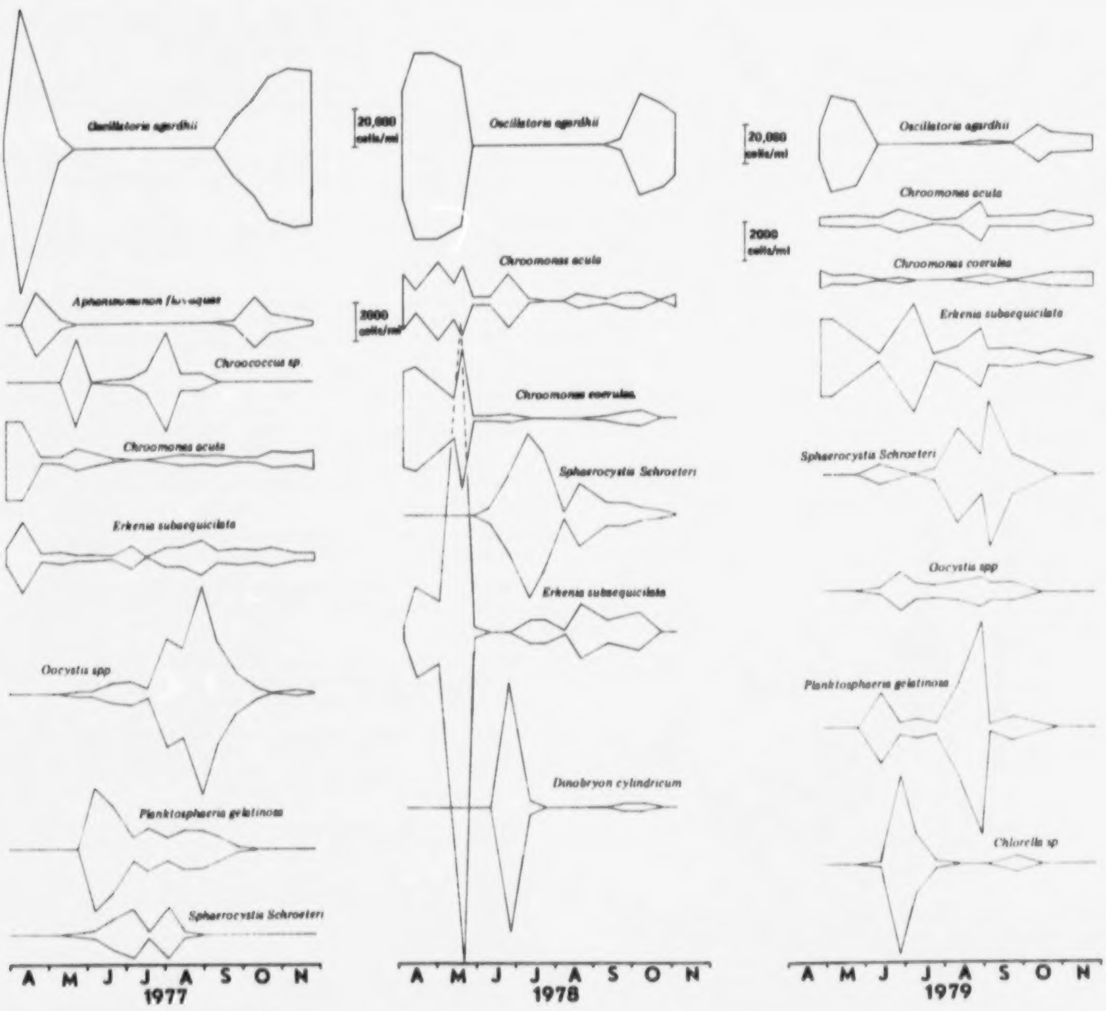

Figure 4. — The seasonal succession of major phytoplankton species, 1977, 1978 and 1979 for Mirror Lake at 2.5 meters.

Table 1. — Morphometric data for Mirror and Pickerel Lakes.

	Mirror Lake	Pickerel Lake
Surface area	5.1 ha	21.0 ha
Maximum depth	13.1 m	4.7 m
Mean depth	7.8 m	2.4 m
Relative depth	2.28%	0.46%
Volume	$4\times10^5 m^3$	$5\times10^5 m^3$
Hydraulic residence time	4 years	0.63 years

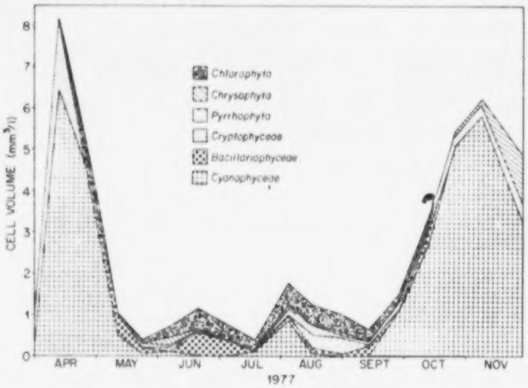

Figure 5a. — Algal biomass in Mirror Lake during 1977 at 2.5 meters.

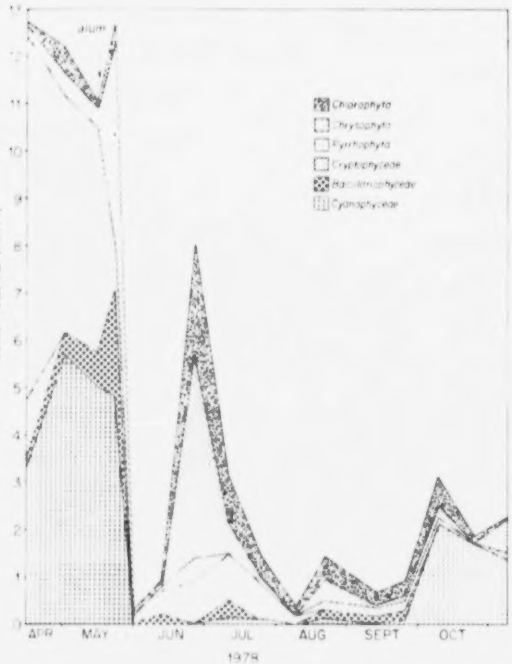

Figure 5b. — Algal biomass in Mirror Lake during 1978 at 2.5 meters. They hypolimnion was treated with aluminum sulfate on May 17, 1978.

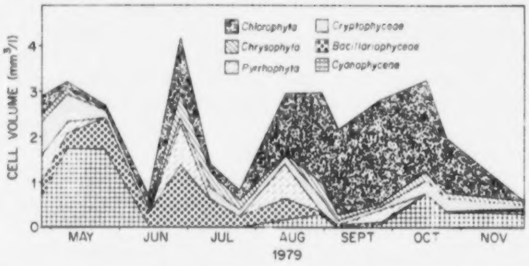

Figure 5c. — Algal biomass in Mirror Lake during 1979 at 2.5 meters.

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HYPOLIMNETIC ALUMINUM TREATMENT OF SOFTWATER ANNABESSACOOK LAKE

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ABSTRACT

Since the 1940's Annabessacook Lake in central Maine has experienced algal blooms resulting from industrial and municipal wastewater inputs. A comprehensive water quality restoration effort combining nutrient diversion, agricultural waste management, and in-lake nutrient inactivation was completed in 1978. The nutrient inactivation technique used aluminum sulfate and sodium aluminate in combination to precipitate phosphorus in the hypolimnion. The use of the two chemicals was necessary to provide sufficient buffering capacity to mitigate potential pH shifts and aluminum toxicity in the low alkalinity (< 20 mg/l) water of Annabessacook Lake. A segregated dual injection system was designed capable of delivering the chemicals simultaneously to any depth between 0 and 7 meters. The hypolimnetic treatment, covering approximately 121 hectares, was completed in a 3-week period. Monitoring data showed a 50 percent decrease in hypolimnetic P approximately 1 month after the application. One year after the aluminum treatment, water quality as measured by total phosphorus, chlorophyll *a*, and Secchi disk visibility, had significantly improved, thus giving rise to optimism about the future of the lake.

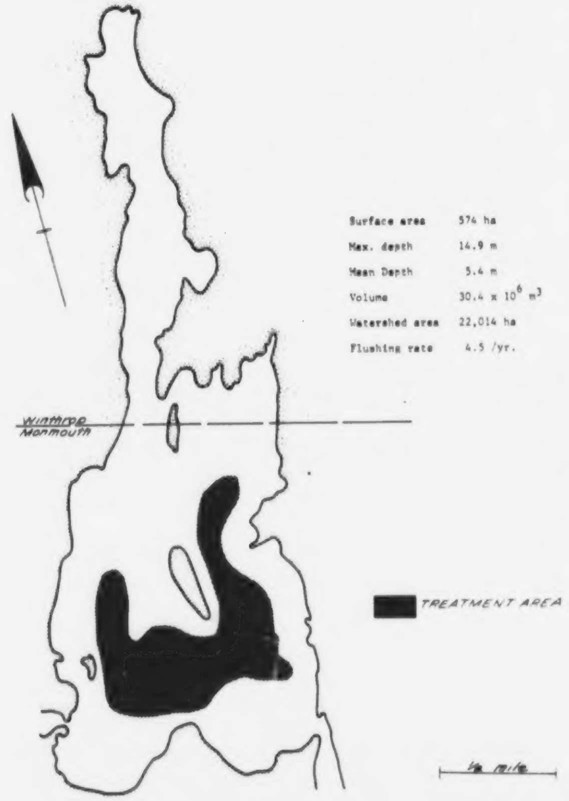
INTRODUCTION

Nutrient inactivation by chemical precipitation is a phosphorus removal technique which has recently been applied to lake restoration. The precipitation agent which has received the most attention is aluminum, although iron, calcium, rare earth metals, and fly ash have also been investigated (Higgins, et al. 1976; Peterson, et al. 1976). Aluminum has often been used in cases where nutrient recycling from bottom sediments would otherwise prolong the recovery of eutrophic lakes long after external sources have been reduced (Knauer and Garrison, 1979; Cooke, et al. 1977).

Aluminum reacts in water at various pH's to stoichiometrically combine with phosphorus to form $AlPO_4$, or undergo hydrolysis to form an amorphous floc which physically sorps soluble phosphorus. Aluminum offers the advantages of low toxicity, effectiveness within the pH range of most natural waters, and is inert to changing redox potentials.

BACKGROUND

Annabessacook Lake is a large (574 hectares) lake located in central Maine (Figure 1). The lake has experienced blue-green algal blooms since the 1940's because of combined discharge of municipal and industrial wastewater into the system. In 1972, an estimated 80 percent of the external phosphorus load to Annabessacook Lake (Scott, 1977), was diverted from the watershed with the construction of a regional wastewater collection system. In 1976, the remaining point sources to the lake were similarly diverted. Despite the elimination of these nutrient sources, however, the lake continued to experience blooms.



Surface area	574 ha
Max. depth	14.9 m
Mean Depth	5.4 m
Volume	30.4×10^6 m^3
Watershed area	22,014 ha
Flushing rate	4.5 /yr.

Figure 1. — Annabessacook Lake.

A 208 Water Quality Management Study (Sage and Moran, 1977) identified agricultural runoff from area farms and internal loading from the lake sediments as the primary reasons for the continued blooms. Based on that study, the Cobbossee Watershed District applied for and received a 314 lakes restoration grant from the U.S. Environmental Protection Agency. The agricultural phase of the restoration project involved developing and implementing agricultural waste management plans for three farms representing 90 percent of the animal units in the watershed. The waste management plans centered around the construction of manure storage facilities, to eliminate the need for winter manure spreading. This is expected to significantly reduce the spring runoff, and thereby, the phosphorus loading to the lake. For further information on the agricultural phase of this project see the paper by Gordon (1980) in this volume.

FEASIBILITY OF ALUMINUM

Preliminary studies were conducted to determine the feasibility of an aluminum treatment in Annabessacook Lake. A seasonal phosphorus budget was developed to verify the significance of internal loading, and a series of laboratory tests were performed to determine if aluminum compounds could effectively tie up phosphorus in Annabessacook water without causing adverse ecological impacts.

Phosphorus Budget

Data were obtained for a phosphorus budget by collecting daily total phosphorus samples, and flow measurements on each of the five tributary streams to the lake, as well as the outlet. Daily lake level measurements were made, and lake phosphorus profiles (1- to 2-meter intervals) were obtained at least biweekly. Precipitation was measured at a site within 1,000 meters of the lake, and its phosphorus concentration determined. Estimates of phosphorus inputs from overland flow and ground water were taken from previous studies (Sage and Moran, 1977; Prescott and Attig, 1977).

The phosphorus budget (Table 1) showed that between June 10 and September 15 approximately 225 kilograms of phosphorus entered the lake from external sources, while 480 kilograms left via the outlet. During the same period, the in-lake phosphorus mass increased by 1,500 kilograms. Inserting these figures into a mass balance formula:

$P_{int} = P_{in\text{-}lake} - (P_i - P_o)$

where P_{int} = internal phosphorus loading

$P_{in\text{-}lake}$ = change in phosphorus mass in the lake

P_i = sum of phosphorus inputs from external sources

P_o = phosphorus loss from the lake via the outlet

yielded a value of 1,800 kilograms of phosphorus attributable to internal sources. Phosphorus concentrations during the open water season ranged from 17 µg/l throughout the lake at spring overturn, to late summer epilimnion and hypolimnion maxima of 72 and 950 µg/l, respectively. For the study period, the epilimnetic (6.5 meters and above) phosphorus increased approximately 550 kilograms (from 590 to 1,140 kg), while the hpyolimnetic phosphorus increased 950 kilograms (from 150 to 1,100 kg) during the same period. These data clearly indicated that internal phosphorus loading was significant in Annabessacook Lake.

Table 1. — Annabessacook Lake summer Phosphorus budget, June 10 — Sept. 15, 1977.

Input	Phosphorus (kg)
Tributaries	153
Precipitation	14
Overland flow	50
Ground water	24
	241
Output	
Outlet Stream	481
In-Lake	
Epilimnion (+)	482
Hypolimnion (+)	1,052
	1,540

Laboratory Studies

A review of the literature showed that aluminum compounds effectively remove dissolved phosphorus and particulate matter from lake water, and that the settled floc retards phosphorus release from bottom sediments. Peterson (1977) and Browman, et al. (1973) found that aluminum is most effective in removing phosphorus from water when the phosphorus is in the inorganic form. Conditions which favor a high inorganic P to organic P ratio are therefore more likely to respond to aluminum treatment. Peterson (1977) also found that phosphorus release from bottom sediments in dimictic lakes remained suppressed several years after treatment with aluminum. Cooke, et al. (1978), and Kennedy (1977), obtained similar results in both Dollar and West Twin Lakes. Where the control of phosphorus release from bottom sediments is an objective, these investigators advocate using a maximum aluminum dosage. They define maximum dose as that dose above which the dissolved aluminum concentration exceeds 50 µg Al/1, a concentration found by Everhart and Freemen (1973) to be safe for rainbow trout.

Filter alum ($Al_2SO_4 \; 14H_2O$), the most commonly used inactivating agent, destroys 0.5 mg/1 of alkalinity for every 1 mg/1 of Al (Sawyer and McCarty, 1967). When alum is used in low alkalinity lakes such as Annabessacook (15 to 20 mg/1 alkalinity as $CaCO_3$) a balancing alkaline compound must be used. An effective means of obtaining this balance is with sodium aluminate ($Na_2Al_2O_4 \; xH_2O$), a high alkalinity compound that offers the advantage of additional aluminum as well as buffering capacity.

To determine the specific responses of aluminum in Annabessacook water, a series of lab tests were run. Table 2 shows the results of the alum/aluminate ratio testing. Ratios yielding solutions with pH's in the 6 to 7 range had "dissolved" (.45μ filtered) aluminum levels below the detection limit of .08 mg Al/1. At pH's outside this range, dissolved Al concentrations rose to potentially toxic levels. As a result of these tests and others, a volumetric alum/aluminate application ratio of 1:1.6 was chosen.

Table 2. Alum-aluminate ratio testing.

Sample	Initial Aluminum (mg/l)	Alum: Aluminate volumetric ratio	pH	Residual Aluminum (mg/l) .45μ filtered
A1	50	1:1	4.5	15
A2	50	1:1.8	5.1 — 5.2	BDL*
A3	50	1:3	8.0 — 8.2	.24 — .47
B1	20	1:1	4.8	1.5 — 1.6
B2	20	1:1.8	6.2 — 6.4	BDL
B3	20	1:3	7.8 — 8.0	.25 — .40
C1	10	1:1	5.2 — 5.3	.09 — .17
C2	10	1:1.8	6.4	BDL
C3	10	1:3	7.1 — 7.2	BDL

*BDL — below detection limit

Another set of tests was used to determine the efficacy of P removal by Al^{3+} over a range of Al^{3+} and P concentrations. Table 3 shows inorganic phosphorus removal exceeded 98 percent for all aluminum dosages.

Table 3. — Phosphorus removal alum aluminate ratio 1:1.6.

Aluminum (mg/l)	Initial P (μg/l)	Final P (μg/l) .45μ filtered	pH
50	500	2	6.4 — 6.6
50	1000	2	6.5
20	500	2	6.5 — 6.6
20	1000	2	6.5 — 6.8
10	500	2	6.5 — 6.7
10	1000	2	6.6 — 6.7
5	500	2	6.5 — 6.8
5	1000	5-6	.67 — 6.8

In addition to chemical tests, bioassays using fish and macroinvertebrates as test organisms were run in an attempt to identify potential toxicity resulting from aluminum application. *Chironomus plumosus,* one of only two benthic species found in the Annabessacook hypolimnion during summer months, was placed in 500 milliliter flasks containing anoxic lake water and 2.5 centimeters of bottom sediment. Aluminum compounds were added on a sediment areal basis in proportion to the maximum dosage anticipated for the project. This amount of aluminum (70,000 mg Al/m^2) formed a thick layer of white floc in the test containers. The flasks were viewed daily, and the test organisms were observed to be lying on the floc, and moving through it as they might through any natural flocculant substrate. Mortality increased with test duration, exceeding 50 percent by day 30. In all cases, however, the controls displayed higher mortality than the test flasks (Table 4).

Table 4. Macroinvertebrate bioassay (*Chironomus plumosus*).

Time	DO	pH	Alkalinity	Temp (°C)	No. Alive	No. Dead	Survival %
0	0.6	6.7	62	9.0			
4 day:							
test	0.2-0.8	5.0	8	9.0	24	1	96
control	0.2-0.3	6.7	65	9.0	15	2	88
15 day:							
test	0.4-0.5	5.8	16	9.0	13	11	54
control	0.2-0.4	7.0	79	9.0	7	7	
30 day:							
test	0.2	5.9	14	9.0	10	13	43
control	0.2-0.4	6.5	78	9.0	2	7	22

In addition to the macroinvertebrate tests, 96-hour static bioassays using golden shiners (*Notemigonus crysoleucus*, a lake inhabitant), were conducted. The tests were carried out in 1-gallon glass jars, each containing two fish in 3 liters of lake water. Test results (Table 5) show no mortality over the entire range of aluminum concentrations. The fish did not appear to be stressed by the test conditions, nor did they avoid the aluminum floc even in the 100 mg Al/l jars.

These tests indicated that by careful manipulation of alum/aluminate ratios and dosages, phosphorus can be effectively removed from lake water. At the same time pH levels could be held within an acceptable range to minimize potential toxicity problems.

Table 5. — 96-hour static bioassay (*Notemigonous crysoleucas*).

Aluminum (mg/l)	pH (0-hour)	pH (96-hour)	% Survival at 96 hours
0	6.8	6.9 - 7.0	100
1	6.7 - 6.9	6.9 - 7.0	100
10	6.9 - 7.0	7.0 - 7.1	100
100	7.1	7.0 - 7.1	100

APPLICATION

From the phosphorus budget and lab test results, it appeared that an aluminum treatment was a feasible restoration technique for Annabessacook Lake. The goal of the in-lake restoration phase was to mitigate internal nutrient loading so that natural recovery of the lake resulting from reduced external loading might be accelerated.

Internal loading occurs from both oxygenated and anoxic bottom sediment. (Lee, et al. 1977). Phosphorus release rates are generally far greater in anoxic sediment, but such conditions are generally confined to hypolimnetic sediment. Low vertical diffusion rates in the metalimnion can severely restrict phosphorus movement from the hypolimnion into the epilimnion, thereby limiting its availability for algal assimilation (Schindler, Hesslein, and Kipphut, 1976; Sweers, 1970). However, significant nutrient transfer can occur in lakes which experience a metalimnetic phosphorus buildup, and are also subject to such phenomena as thermocline migration and/or internal seiches (Stauffer and Lee, 1974; Mortimer, 1971).

Phosphorus from oxygenated epilemnetic sediment, though less rapidly released, is more immediately available for primary production than hypolimnetic released phosphorus, and can be a significant internal source. In smaller lakes, where the littoral zones comprise a relatively large portion of the lake area, littoral inputs may be particularly important, especially where inputs are enhanced by groundwater inflow and macrophyte pumping (Twilley, et al. 1977; McRoy, et al. 1972). Conversely, in larger eutrophic lakes, seasonal nutrient inputs from the hypolimnion may dominate. In most lakes it is likely that a number of internal sources contribute nutrients, at least on a seasonal basis.

Annabessacook Lake has both a large anaerobic hypolimnion and a macrophyte-covered littoral zone. The lake annually shows a large phosphorus buildup concurrent with the loss of oxygen in the hypolimnion. This buildup extends into the thermocline by midsummer at which time it is likely that considerable transfer to the epilimnion occurs. Also the lake has historically experienced fall blooms that correspond with fall overturn when the nutrient-enriched bottom waters become incorporated in the rest of the lake. It was felt that if hypolimnion P was made unavailable, fall blooms might be less severe. The reduction of any phosphorus transferred out of the hypolimnion to the littoral zone during overturn might mean that this phosphorus would not be available through some form of littoral release at a future date. Although littoral zone phosphorus release might be significant in Annabessacook Lake, the vast size of the littoral zone (400 hectares) presented enormous logistical problems.

It was decided that a hypolimnetic aluminum application, covering the entire area of anaerobic sediment, would be most appropriate for Annabessacook. Such an application would accomplish two objectives. First, the application would strip the hypolimnion of phosphorus by precipitation and entrapment. This could be expected to be most effective if done in mid to late summer when hypolimnetic phosphorus concentrations in Annabessacook Lake are greatest, and 95 percent of the hypolimnetic P was an orthophosphate. Second, aluminum floc would chemically seal the sediment, preventing future phosphorus release. Through a hypolimnion application, maximum aluminum concentration could be achieved in the area of greatest release. Aluminum application dosages in the top meter of treated water were 25 and 34 mg Al/1 for areas 7 to 10 meters and over 10 meters deep, respectively.

The simultaneous placement of two chemicals over an area of 150 hectares at a depth of 7 meters presented a sizable logistics problem. Because of the large amounts of time and travel required to resupply, a single large capacity vessel rather than a number of smaller vessels was chosen for the application. Both the commercial aluminum sulfate and sodium aluminate were obtained in liquid form for ease of handling. The chemicals were delivered to the lakeside base of operations at staggered intervals in 3,500 to 4,700-gallon tank trucks. At the base, the chemicals were temporarily stored in two 1.2 x 5.49 meter diameter polyvinyl-lined swimming pools, erected especially for this project. The pools, with a capacity of 7,600 gallons each, proved to be very adequate holding facilities. The chemicals were pumped from the pools approximately 25 meters to a three-compartment, 23 m3 mild-steel tank truck, mounted on a 12 m x 7.6 m barge (Figure 2). The barge, a series of iron pontoons, was transported from Portland, Maine and placed in Annabessacook Lake by the Maine National Guard.

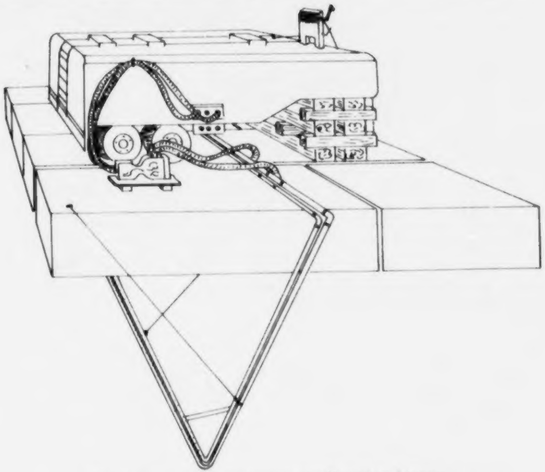


Figure 2. — Aluminum application barge.

The tank truck was valved so that each compartment could deliver its product via pumps to a completely segregated dual diffuser system. It was necessary to maintain complete separation of the chemicals, because contact led to the instantaneous formation of a precipitate which could clog the diffuser. The diffuser, constructed of 5 centimeter diameter black iron pipe, formed a 8.8 m x 7.6 m rectangle with 1.5 m extensions at each end of one of the longer sides. The other 8.8 m side rested in a cradle and straddled the barge approximately amidship, with the elongated side extending out over the bow of the barge. With this arrangement, the diffuser was able to pivot from a horizontal, above-water traveling position, to a vertical applying position, reaching a maximum of 7.5 meters below the water surface. The diffusing pipes were drilled with 6 mm holes every 15 cm over their 11.9 m lengths. The holes of the two pipes were positioned to coincide with one another and were angled to provide good chemical mixing when the system is as operation. The overall diffuser system weighed over 360 kilograms and was raised or lowered by a winch in the center of the barge, and a block-and-tackle on either side. The diffuser was positioned at 7 meters below the water surface during application.

The chemicals were delivered from the tank truck to the diffuser by two 3-horsepower gasoline driven pumps through sections of 5 cm diameter hose. Valving, both on the tank truck and in-line, and flow meters accurately regulated the chemical supply. Dosage rates were coordinated with barge speed and depth of area being treated. Surface trials of the system showed excellent floc formation and even dispersal along the length of the diffuser. The use of quick-connect hoses permitted easy and efficient flushing of the pumps and diffuser at the end of each day's application. Flushing was necessary because of the corrosive nature of the chemicals.

first, these buoys were kept to the outside of the barge, and the previously dropped buoys were picked up by trailing boats. This insured that the entire area would be covered, using a minimum number of buoys.

The aluminum application took approximately 18 days, averaging 10 hours per day for a standard crew of five persons. In addition, four to five local volunteers manning two to three boats were necessary to coordinate buoy placement and pickup.

There was a considerable amount of down-time during the application phase due to engine failures, tank leaks, and damage to diffusers. Despite these problems, approximately 95 percent of the area originally targeted for treatment received treatment.

The barge was propelled by two powerboats (75 horsepower and greater), one on each side toward the stern of the barge. Steering was accomplished by varying the engines' speeds, and/or direction, of thrust. This combination provided excellent maneuverability. The barge was able to maintain a speed of 1 to 2.5 miles per hour under moderate wind conditions.

Aluminum treatment was carried out where depths exceeded 8 meters. The treatment route generally followed a pattern of decreasing concentric circles. To distinguish treated areas from untreated areas, buoys were dropped off the inside of the barge every 100 meters. On the next pass, a smaller circle inside the

POST APPLICATION RESULTS AND CONCLUSIONS

The hypolimnetic aluminum treatment, combined with the agricultural waste management controls, dramatically reduced phosphorus content in Annabessacook Lake in 1979 (Figures 3 and 4). The maximum phosphorus mass in the lake in 1979 was 1,030 kilograms, compared to over 2,200 kilograms in 1977, a reduction of greater than 50 percent. Internal recycling in 1979 contributed 625 kilograms phosphorus from spring overturn to mid-August when the phosphorus mass in the lake reached its highest level. This represents a 65 percent reduction from the 1,800 kilograms attributed to internal loading in 1977.

Phosphorus decreased in both the epilimnion and the hypolimnion in the post-application year. The disparity between the 2 years' concentrations became increasingly great, especially in the hypolimnion, as the summer proceeded. Over the summer, the seasonal increase in hypolimnetic phosphorus mass was only a third of the 1977 increase, 320 kilograms compared to 1,100 kilograms, despite similar temperature and dissolved oxygen conditions for the 2 years. The implication is that the aluminum floc effectively sorbed sediment-released phosphorus. The increase in hypolimnetic phosphorus that did occur might have been caused either by some fugitive sediment-released material, or mineralization of sedimented algal cells raining down on the floc.

Epilimnetic phosphorus levels were also reduced in 1979. This was especially true in the early summer, when post-treatment concentrations showed very little increase. Only later in the summer, when weather conditions facilitated phosphorus transfer from the hypolimnion did epilimnetic phosphorus concentrations rise.

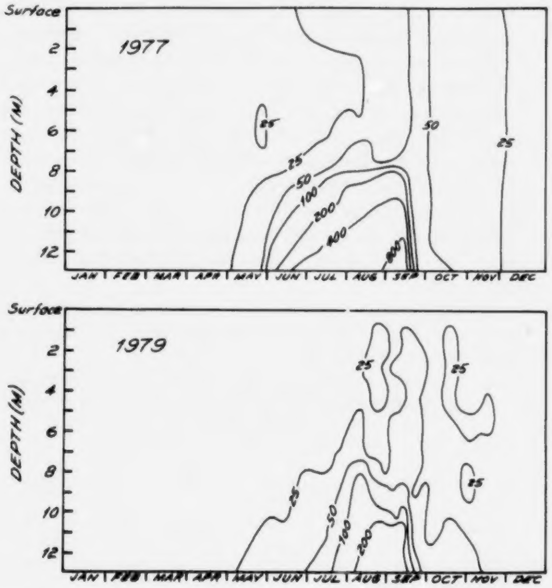


Figure 3. — Total phosphorus isopleths in Annabessacook Lake, 1977 and 1979.

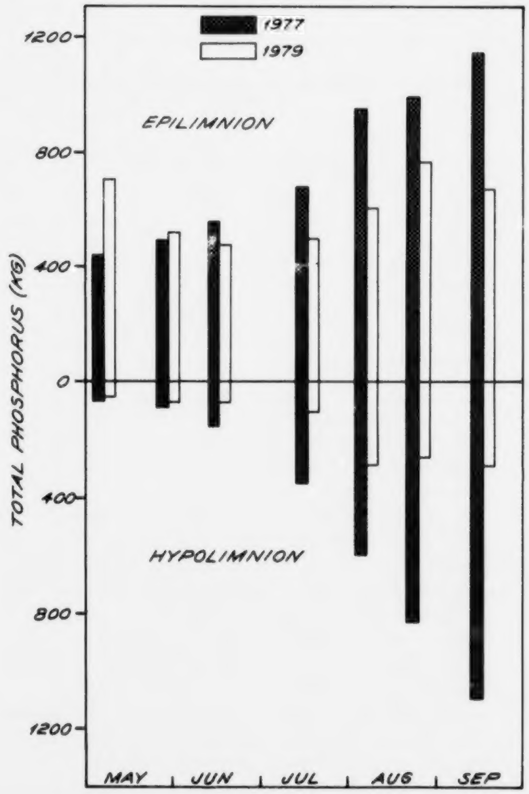


Figure 4. — Total phosphorus mass in Annabessacook Lake, 1977 and 1979.

The reasons for the reduced phosphorus levels in 1979 are not completely understood. It seems likely that they center around the aluminum treatment, although climatological conditions and reduced external loading from agricultural lands may also have contributed. It appears that over the summer the aluminum floc was effective in sorbing internally regenerated phosphorus, thereby suppressing hypolimnetic levels, and eventual transfer to the epilimnion. In addition, littoral loading may also have been reduced following the movement of phosphorus-binding aluminum to the littoral zone during fall and spring overturns.

Whatever the reasons, the dramatic results were also manifested in the Secchi disk and chlorophyll *a* data (Tables 6 and 7).

Table 6. Annabessacook Lake Secchi disk visibility (meters) monthly means.

	1975	1976	1977	1978	1979
May	4.1	3.1	3.2	3.0	3.5
June	4.2	3.7	2.0	3.1	3.9
July	2.6	2.2	1.2	1.1	3.4
August	2.1	2.9	1.1	1.8*	2.6
September	2.1	1.5	2.3	1.8	3.8
October		2.4	1.9	2.5	3.9
Mean	3.0	2.6	2.0	2.2	3.5

* Aluminum Application

Table 7. — Annabessacook Lake chlorophyll *a* (μg/l) monthly means.

	1976	1977	1978	1979
May	5.4	11.5		4.7
June	4.5	11.5	6.2	4.7
July	6.2	18.7	29.3	6.2
August	6.8	8.7	23.6*	12.7
September	14.9	24.4	17.2	8.7
October	15.6	17.8	19.0	7.4
Mean	8.9	15.4	19.1	7.4

* Aluminum Application

Especially encouraging was the absence of a fall bloom which in the past has been stimulated by the incorporation of nutrient-rich hypolimnetic water into the epilimnion.

The summer of 1979 was the first since records have been kept in which Secchi disk visibility was always better than the Maine-designated bloom level of 2 meters. A review of the restoration progress made on Annabessacook Lake resulting from watewater diversions, agricultural waste management, and the aluminum treatment, and reflected by changes in Secchi disk visibility is presented in Table 8.

To date the results of the aluminum application are very encouraging, but the degree of success cannot be judged until sufficient data are available.

It can be stated though, that the Annabessacook Lake Restoration project has shown that a large scale hypolimnetic aluminum application on a soft-water lake is a feasible lake restoration technique.

Table 8. — Secchi disk visibility days:*

Secchi disk Visibility (Meters)	1972 (Before sewage diversion	1977 (Before lake restoration project)	1979 (After aluminum treatment and agriculture controls)
0 — 0.9	43 days	10 days	0 days
1 — 1.9	63 days	67 days	0 days
2 — 2.9	0 days	28 days	30 days
3 — 3.9	0 days	1 day	40 days
4 — 4.9	0 days	0 days	35 days
5+	0 days	0 days	1 day

* based on a summer season from June 1 — Sept. 15.

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MEDICAL LAKE IMPROVEMENT PROJECT: SUCCESS STORY

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ABSTRACT

Medical Lake is an alkaline lake in Eastern Washington State that has historically exhibited nuisance algal blooms, extensive summer anoxia, and high nutrient concentrations. The lake is located within the corporate limits of the town of Medical Lake and its eutrophic condition resulted primarily from internal phosphorus cycling. The lake lies in a closed basin with a drainage area of 3.5 km^2. Land use is predominantly residential. Muncipal sewage collection and treatment began in 1964. The lake was treated with alum during August and September 1977, with dramatic results. Total phosphorus and orthophosphorus concentrations have declined from over 400 and 300 $\mu g l^{-1}$ to less than 60 and 3 $\mu g l^{-1}$, respectively. Chlorophyll *a* has remained below 5 $\mu g l^{-1}$, and often is undetectable. Secchi disk depths have averaged 5 m and ranged from 2.5 to 11 m. Before treatment Secchi depths averaged 1.2m. The extent of summer and winter anoxia has declined and increased oxygen levels have improved the fishing habitat. Fifteen thousand 6.3 cm rainbow trout fingerlings were planted in the lake during June 1978. These fish currently exceed 1 kg in weight and 50 cm in length. Recreational use of the lake and shore front park has increased accordingly. Activities include swimming, boating, and picnicking. Furthermore, public fishing is expected to be permitted during 1981.

INTRODUCTION

Medical Lake is a freshwater lake located near the town of Medical Lake, approximately 24 kilometers southwest of Spokane, Wash. For several decades, a high phosphorus concentration contributed to the recurrence of algal blooms and floating mats of algae in the lake. Along with the thick algal surface scum, offensive odors were associated with decaying algae and hydrogen-sulfide-laden bottom waters. These conditions allowed only limited use of the lake.

To improve recreation, the town of Medical Lake sponsored a project to restore the water quality of the lake. The project was directed by Battelle, Pacific Northwest Laboratories, with major support from Eastern Washington University. Financial support came from the U.S. Environmental Protection Agency, the State of Washington Department of Ecology on a matching basis, the town of Medical Lake, and Spokane County. The objectives of the project were to reduce phosphorus and algae levels, increase oxygen levels, and improve water clarity to permit recreational use of the lake and possibly establish a fishery.

Data collection and laboratory analyses showed that the high phosphorus concentration came from an internal phosphorus cycle. Consequently, it was decided that the best method for improving water quality would be to disrupt the cycle. Of the procedures available, phosphorus inactivation by chemical precipitation appeared to be the most effective and economical method. The technique chosen for phosphorus inactivation consisted of multiple applications of aluminum sulfate (alum).

The project began in June 1977. The alum was applied over a 5-week period beginning in August 1977. Water quality monitoring was conducted prior to, during, and after the application. Preliminary results reported by Gasperino, et al. (1978) indicated that the treatment had reduced phosphorus and algae concentrations and increased water clarity. Water quality monitoring continued until June 1980.

This paper presents results of the restoration project through December 1979. Detailed results are available in the project's final report (Gasperino, et al. 1980). Additional data on methodology development can be found in the preliminary report (Gasperino, et al. 1978). A fisheries report which includes water quality data through 1980 will be prepared during 1981.

LAKE HISTORY AND CHEMISTRY

Medical Lake lies in a closed basin within the corporate limits of the town of Medical Lake. The basic physical characteristics of the lake are as follows:

Area	158 acres	64 ha
Volume	5,026 acre-ft	$6.2 \times 10^6 m^3$
Maximum Depth	60 feet	18 m
Mean Depth	33 feet	10 m
Maximum Length	5,600 feet	1.7 km
Maximum Width	1,300 feet	0.4 km

The basin was scoured from basalt of the Columbia River group by recurring glacial floods. The largest flood occurred between 18,000 and 20,000 years ago. Land use in the drainage area (3.5 km^2) is predominately residential. Approximately 36 percent of the shoreline length (5.0 km) is developed. Municipal sewage collection and treatment have been operational since 1964. Prior to 1964, septic tanks and cesspools were employed.

A familiar vertebrate inhabitant of the lake is the painted turtle (*Chrysemys picta*). The only known fish populations to inhabit the lake prior to the alum treatment were small stocks of tench (*Tinca tinca*) and carp (*Cyprinis carpio*). Tench still inhabit the lake. Since the completion of the alum treatment approximately 30,000 rainbow trout (*Salmo gairdneri*) have been introduced to the lake over 3 successive years.

Bauman and Soltero (1978) described the limnology of Medical Lake in detail in 1974. Their study showed a high concentration of phosphorus. Furthermore, they concluded that most of the phosphorus was being recycled within the lake. Pretreatment surveys during the restoration project confirmed these earlier results.

Prior to treatment, the major sources of phosphorus within the lake were decomposing algae and bottom sediment, which released the nutrient throughout the summer. Phosphorus was then mixed throughout the lake during the fall. Thus, the algal production of one growing season stimulated algal growth during the following growing season. Very little phosphorus probably enters the lake from the surrounding basin because the lake receives no known sewage effluent or agricultural runoff and has no surface inlets.

LABORATORY ANALYSES

After analyzing the lake's characteristics, studying the literature, and reviewing other lake restoration techniques, alum treatment was selected as the most appropriate method for inactivating the phosphorus in Medical Lake. Previous literature on eutrophication indicated that an 87 percent orthophosphorus reduction was probably necessary to reduce algal blooms in Medical Lake.

Before the alum was applied, laboratory analyses were made to determine the quantity rate of application, and type of mixing required. The following requirements were necessary to achieve an 87 percent reduction in orthophosphorus*:

1. A whole-lake alum ($Al_2(SO_4)_3 \cdot 18H_2O$) concentration of 150 mg l^{-1};
2. Vigorous mixing of the alum as a liquid slurry rather than as dry crystals;
3. Multiple doses rather than a single dose; and
4. Combined surface and subsurface applications rather than surface applications alone.

*. Orthophosphorus is soluble reactive phosphorus. For the laboratory analyses, orthophosphorus was measured as an indication of overall phosphorus reduction.

Detailed results of these tests are presented in the preliminary and final reports (Gasperino, et al. 1978, 1980).

ALUM DISPENSING SYSTEM

Once the parameters required for treatment were determined, a dispensing system was needed to provide a fast, safe, and efficient means of placing the alum into the water. Two pontoon barges were used to distribute alum in a well-mixed, uniform concentration at prescribed depths: a 12-meter barge for deep areas, and a 8.5-meter barge for shallow areas. Each barge was equipped with tanks for carrying the alum, a distribution pump, and an injection manifold. The injection manifold allowed alum distribution at the surface or 4.5 meters.

The time for dispensing a load of alum varied depending on the type of application. Subsurface injection took about 45 minutes with the 12-meter barge and 25 minutes with the 4.5-meter barge. Surface applications were faster. The barges could increase speed because less manifold drag occurred and, consequently, the pumping rate was increased to keep the volume of alum dispensed per dispensing zone constant. Figure 1 illustrates the application barge.

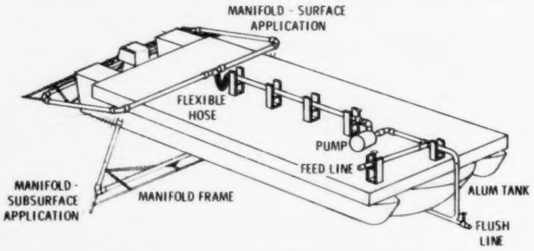
MANIFOLD - SURFACE APPLICATION
FLEXIBLE HOSE
PUMP
FEED LINE
MANIFOLD - SUBSURFACE APPLICATION
MANIFOLD FRAME
ALUM TANK
FLUSH LINE

Figure 1. — Alum dispensing system.

For the alum application, the lake was divided into six equal zones with marker buoys to facilitate a systematic distribution. The barge pilots treated each zone in a series of back and forth passes orienting themselves by the marker buoys and landmarks on shore. The sequence of passes was: two subsurface applications, two surface applications, two more subsurface applications, then one surface and one subsurface application. The second application in a zone was not made until all other zones had been treated.

RESULTS

The results of the water quality monitoring through June 1980 show that the alum treatment was highly successful in decreasing phosphorus levels, eliminating algal blooms, and increasing water clarity. Thirty-two sampling cruises were completed between January 17, 1978 and December 10, 1979. Biweekly samplings were made June through September and at monthly intervals for the balance of the study. One station, at the deepest point, was sampled throughout the project. Water samples were taken at 2-meter intervals from the surface to the bottom with a 1-liter Kemmerer sampler. Also, a euphotic zone composite was collected by combining samples (usually taken at 1 meter intervals) of equal volume from the surface to the lower limit of the euphotic zone.

Complete profiles of chemical and biological parameters are presented in the final report (Gasperino, et al. 1980). Figures 2 through 6 illustrate average lake concentrations of important water quality data. Most of these data indicate that a substantial improvement has occurred as a result of the treatment.

Figure 2 shows the mean monthly total and orthophosphorus concentrations from December 1976 through December 1979. Prior to the alum treatment, the mean monthly total and orthophosphorus concentrations were approximately 0.47 and 0.31 mg l^{-1}, respectively. Concentrations for both fractions declined immediately following treatment, but it was not until fall turnover that the impact of the alum application was fully realized. A comparison of overall mean total and orthophosphorus concentrations, before October and after November 1977, showed that each fraction decreased approximately 87 and 97 percent, respectively.

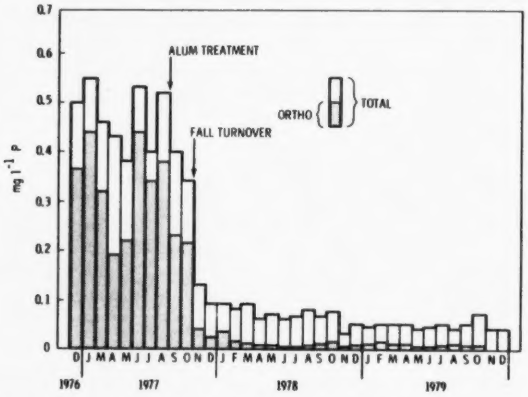

Figure 2. — Mean Monthly Total and Orthophosphorus Concentrations (mg 1^{-1} P) Before, During and Following Treatment.

The cause for the substantial reduction of phosphorus at fall turnover, particularly during 1977, is not clearly understood. A possible explanation of the decline could be that the sedimented floc still possessed phosphorus sorptive properties. Therefore, additional phosphorus removal was effected by circulation of the entire water mass. This mechanism might also explain why phosphorus concentrations continued to decline in 1978 and 1979.

Figure 3 presents the mean monthly chlorophyll *a* values for all months of study. The overall mean chlorophyll *a* concentration prior to and during the alum treatment (December 1976 to September 1977) was approximately 25.2 mg m^{-3}. The overall mean concentration for 1978 and 1979 was 3.20 mg m^{-3}, a decrease of 87 percent. Before treatment the maximum mean monthly chlorophyll *a* concentration was 59.8 mg m^{-3} in May of 1977, while the maximum post treatment value was 17.5 m^{-3} in February 1978. Mean growing season (May-September) values for chlorophyll *a* concentrations during 1977, 1978, and 1979 were 16.7, 2.19, and 2.50 mg m^{-3}, respectively.

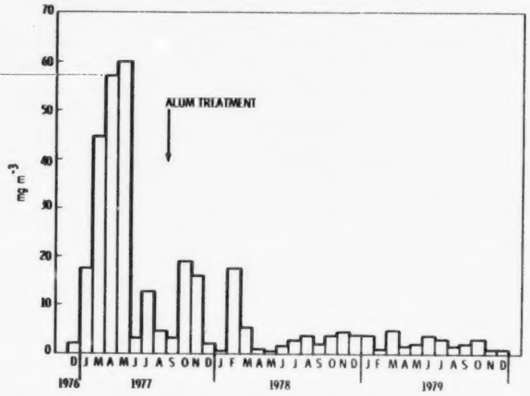

Figure 3. — Mean Monthly Chlorophyll *a* Concentrations (mg m^{-3}) Before, During and Following Alum Treatment.

Figure 4 shows the mean monthly dissolved oxygen concentrations for the water column before (December 1976 to July 1977), during (July 1977 to September 1977) and following (October 1977 to December 1979) the alum treatment. Rather large fluctuations in concentration were evident before and during the treatment. Following treatment, variation in mean monthly oxygen concentrations decreased with levels tending to be near 5 mg 1^{-1}. Decomposition of the sedimented organic matter, resulting from the treatment, has probably negated significant improvement in the overall dissolved oxygen regime of the lake. In the future, overall oxygen concentrations should significantly increase with the stabilization of the sedimented materials.

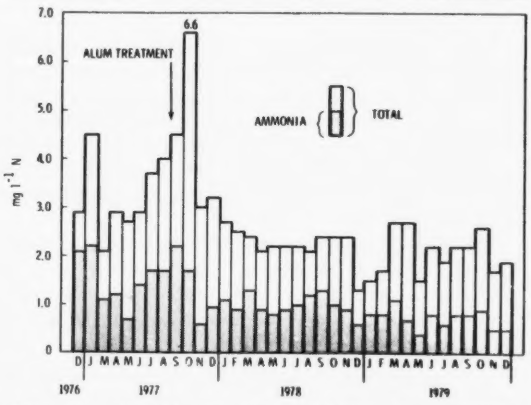

Figure 4. — Mean Monthly Dissolved Oxygen Concentrations (mg 1^{-1}) Before, During and Following Alum Treatment.

Figure 5 presents the mean monthly total and ammonia nitrogen concentrations before, during, and following the alum application. Prior to the alum treatment the total nitrogen concentrations for the water column approached 3.2 mg l^{-1}; however, post-treatment to to 0.9 mg l^{-1} following treatment (about a 40 percent decrease).

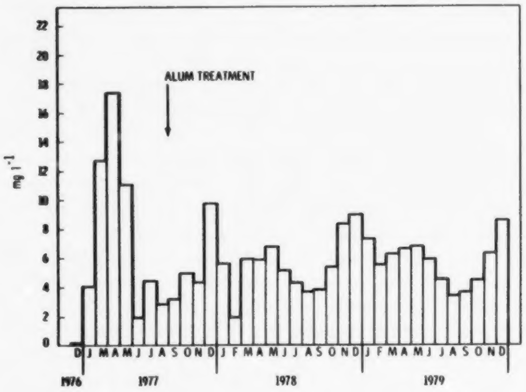

Figure 5. — Mean Monthly Total and Ammonia Nitrogen Concentrations (mg l^{-1} N) Before, During and Following Alum Treatment.

Water clarity has significantly improved since the alum treatment (Figure 6). The overall mean Secchi disk visibility before the treatment was 2.4 meters; after the treatment it was 4.9 meters. The improved water clarity is a tangible indication of reduced algal growth.

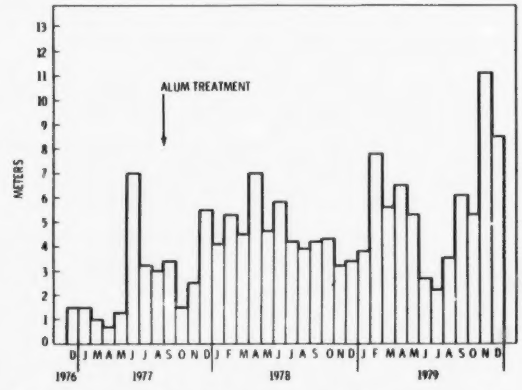

Figure 6. — Mean Monthly Secchi Disk Visibilities (meters) Before, During and Following Alum Treatment.

In 1977, the phytoplankton community was dominated by the Cryptophyceae and Cyanophyceae (Soltero, et al. 1978). The Cryptophyceae reached a maximum standing crop of 20.52 mm^3 l^{-1} in May 1977 while the Cyanophyceae, primarily *Microcystis aeruginosa*, reached bloom proportions August 3 just prior to the alum treatment. Since the treatment, the Chlorophyceae and Cryptophyceae have dominated the phytoplankton community. In 1978, the chlorophyceaen standing crop steadily increased from a low of 0.09 mm^{3}l^{-1} in February to a seasonal high of 3.17. The maximum cryptophyceaen standing crop occurred in March during both study years, 1.57 mm^{3}l^{-1} and 2.01 mm^{3}l^{-1} in 1978 and 1979, respectively. Except for a small blue-green pulse in the summer of 1978, primarily consisting of *Synechocystis* sp. and *Oscillatoria tenuis*, the mean monthly cyanophyceaen standing crop during both study years was less than 0.07 mm^{3}l^{-1}. Contributions to the total cell volume by the remaining phytoplankton classes were minimal during both years.

In 1977, a shift in the primary growth limiting nutrient (to *Selanastrum capricornutum* as determined by algal assay) from nitrogen to phosphorus occurred following the alum treatment (Soltero, et al. 1978). Algal assay results for 1978 show that phosphorus continued to limit the growth of *Selenastrum* throughout the study year. The extent of phosphorus limitation during 1978 was also evident by the overall high total inorganic nitrogen to orthophosphate ratio (approximately 50:1) determined in the euphotic zone.

Keizur (1978) suggested that the decline and replacement of blue-green algae with more palatable greens and cryptomonads could result in decreased rotifer numbers with a corresponding increase in daphnid and diaptomid density. He proposed that the macroconsumers, such as *Daphnia* and *Diaptomus*, would make up a greater majority of the zooplankton standing crop in response to the phytoplankton shift. He further suggested that if fish stocking was implemented, the larger macroconsumers would provide an excellent food source for the fish.

Since the alum application, a substantial reduction of blue-green algae has occurred with a corresponding replacement by greens and cryptomonads. In response to this change, the rotifer population declined with a corresponding increase in daphnid and diaptomid populations. Thus, a greater proportion of the zooplankton community now consists of macroconsumers.

Medical Lake was stocked with 14,000 rainbow trout fingerlings (*Salmo gairdneri*) in June 1978, 12,000 in June 1979, and an additional 4,000 fingerlings a year later. Preliminary results of an ongoing fisheries study indicate that fish growth and condition are excellent (Knapp, pers. commun.). Some of these 6.3 cm fingerlings have grown greater than 50 cm and weigh more than a kilogram. Stomach analyses of the trout revealed that the larger and abundant macroconsumer, *Daphnia pulex*, has been an excellent food source.

Continuing studies suggest, however, that the fish predation has caused a size reduction in *D. pulex* and affected the appearance and increased the numbers of smaller zooplankters (Mires, 1980). Since modification of the zooplankton community as a result of selective grazing can greatly influence the composition of the phytoplankton standing crop (Brooks and Dodson, 1965), it is important that a large complement of macroconsumers (i.e., *D. pulex*) be maintained in Medical Lake to help minimize algal standing crop, a primary objective of the restoration project. Elimination of large grazers because of excessive fish predation will only shorten the life expectancy of the restoration. Present understanding of the food chain balance would indicate that further fish stocking would be inadvisable at this time. Further study of the trout, zooplankton, and phytoplankton relationships will give more exact information as to the number of fish Medical Lake will support without compromising its improved water quality.

PROJECT COSTS

Total project costs (Table 1) included the cost of alum; labor costs for monitoring alum application, data analysis, and project management; and equipment rental and outfitting. Water quality monitoring and data analysis accounted for a large part of the expenditures. The price of the alum was also significant.

Table 1. — Project costs for Medical Lake restoration.

Alum		$ 90,000
Labor		
	Monitoring (1/77-6/80) (Biological Physical, Chemical)	55,000
	Chemical Application (8/77-9/77)	5,000
	Project Management, Planning, Coordination and Data Analysis (1/77-6/80)	72,000
Equipment		
	Barge Rental	8,000
	Vehicle Rental	1,150
	Pumps, Supplies	7,250
Bond		1,500
	TOTAL	$239,900

CONCLUSIONS

The alum treatment of Medical Lake has significantly improved the lake's water quality. Mean concentrations of total and orthophosphorus steadily declined from January through December during both study years. Mean orthophosphorus concentrations declined 5.5 and threefold in 1978 and 1979, respectively. Total phosphorus concentrations decreased approximately twofold during each study year.

Mean monthly total and ammonia nitrogen levels were lower in 1979 than 1978. Since the alum treatment, mean total and ammonia nitrogen concentrations have declined 22 and 40 percent respectively.

Algal assay results for 1978 showed that phosphorus was the primary growth limiting nutrient to *Selenastrum*. Phosphorus limitation was also evident by the overall high 1978 euphotic zone total inorganic nitrogen to orthophosphate ratio (approximately 50:1).

Chlorophyceae and Cryptophyceae dominated the phytoplankton community during 1978 and 1979 with the Chlorophyceae being the major contributor to the total cell volume. Cyanophyceae, the dominant phytoplankton class in 1977, was a minor contributor to the total cell volume in both study years. The overall mean phytoplankton standing crop ($mm^3 l^{-1}$) before and during the application was reduced by 91 percent when compared to the mean standing crop following treatment. This decline in algal standing crop also related well with the overall 87 percent reduction in mean chlorophyll *a* concentration ($mg\ m^{-3}$) following treatment.

Water clarity has significantly improved since the alum treatment. The overall mean monthly Secchi disk visibility before and after treatment was 2.4 and 4.9 meters, respectively.

Hypolimnetic anoxia was evident during 1978 and 1979 with the periods of anoxia lasting 9 and 5 months, respectively. A sediment oxygen demand, because of the stabilization of organic matter deposited on the lake bottom as a result of the alum treatment, was the probable cause of anoxia during both study years. Any improvement in the dissolved oxygen regime of Medical Lake as a result of the treatment possibly has been negated by this sediment demand. However, the volume of water that became anoxic in 1978 and 1979 was much less than that of 1977.

Heavy selective grazing, caused by recent fish stockings, appears to have effected a reduction in mean body length of *D. pulex* and may have promoted a species composition shift as evidenced by the appearance of other smaller zooplankters.

The improvement in water quality and clarity has increased recreational use of the lake. Activities include swimming, boating, picnicking, and fishing. Property values around the lake have risen, and users of the park have increased on summer weekends from fewer than 100 to an estimated 1,000. As a result, the town is seeking additional funds to improve the park and docking facility. And for the first time, the town has hired a lifeguard.

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DETERGENT MODIFICATION: SCANDINAVIAN EXPERIENCES

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ABSTRACT

In Scandinavia the concern over the role of detergents in water deterioration has been focused on over-fertilization by tripolyphosphates. Agreements between authorities and industry have lowered the phosphate content to about 7 percent as P. Detergents with lower P-content have been based on NTA or soap. Detergents totally free of phosphate have also been manufactured, based on a mixture of adipate-acetate or citrate. A special campaign for phosphate-free detergents is running within the Lake Mjosa catchment area in Norway. The measures taken have reduced phosphorus load on the water bodies, but are difficult to evaluate, as other P-sources also were reduced parallel to the decrease of detergent P. In any case, no harmful effect of NTA has been reported.

INTRODUCTION

In Scandinavia scientific concern about the role of the synthetic detergents in water deterioration started in Sweden at the end of the 1950's. The main interest was focused on the role of the detergent polyphosphates in the over-fertilization of natural waters, the eutrophication process.

During the 1960's the discussions became very public and culminated when advanced wastewater treatment for phosphorus removal became standard during the early 1970's. When the discussions began, however, there was no method for efficient phosphorus removal at the sewage treatment plants. At this time, replacement of phosphorus in the detergents by some other builder — especially different types of chelating agents — seemed to be the most rapid way to start decreasing the P-load on our waters. It was also stated that this was a partial solution and that a more total P-removal was necessary to achieve real improvements, especially in urban waters.

Different measures and modifications resulted, among other things, in lower consumption of detergent phosphorus. In Norway a special campaign for increasing the sale of P-free detergents was found necessary in the Lake Mjosa catchment area. In the other Scandinavian countries little interest has been devoted to the environmental effects of detergents.

DETERGENT MODIFICATIONS IN SWEDEN

The most common commercial synthetic detergents were tested and found to be excellent sources for algal growth, although different inhibitory effects of the other detergent components were noted (Forsberg, Jinneraot, and Davidsson, 1967). Nitrilotriacetic acid (NTA) was considered a good substitute for polyphosphate, stimulating numerous investigations. The first reports on biological degradation of NTA appeared in 1967 (Swisher, Crutchfield, and Caldwell, 1967; Forsberg and Lindqvist, 1967a,b). Since that time many papers dealing with NTA have been published. Comprehensive lists of references have been presented (Monsanto, 1977). NTA has been used in Sweden as a substitute for polyphosphate since 1968. In addition, other modifications and measures have also been made, examples of which are given in Table 1.

Table 1. — Examples of detergent modifications in Sweden.

Compounds	Percent			
	A	B	C	D
Surface active agent	13.0	14.0	3.0	22.0
Sodium tripolyphosphate	30.0	9.0	7.0	-
Soap	-	-	38.0	-
NTA	-	12.0	-	-
EDTA	0.5	-	-	4.0
Sodium-adipate	-	-	-	39.0
Sodium-acetate	-	-	-	
Sodium-carbonate	18.0	31.0	16.0	-
Sodium-silicate	8.0	5.0	8.0	-
CMC	1.0	1.6	1.0	2.0
Perborate	18.0	20.0	15.0	-
Mg-silicate	1.0	1.2	-	4.0
Optical whiteners	0.3	0.3	0.4	0.5
Sodium-sulfate	1.5	1.5	-	27.0
Perfume	0.3	0.2	0.3	0.1
H_2O	9.0	4.0	11.0	1.4

The surface active agents earlier causing well-known water pollution problems were changed to more biologically degradable ones. This was finalized in 1969 for anionic tensides and in January 1973 for non-ionic ones.

An agreement between the National Environment Protection Board and industry, limited the phosphate content to a maximum of 7.5 percent P or 30 percent as sodiumtripolyphosphate, and to 10 percent P in machine dishwashing agents. Detergents with lower P-content have been based on NTA or soap (Table 1, B and C). Totally phosphate-free detergents have also been manufactured, based on a mixture of adipate-acetate (Table 1, D) or citrate.

Through these measures the phosphate amount in household detergents was reduced from 4,100 tons in 1968 (calculated as P) to about 3,000 in 1970, corresponding to 28 percent. After that the detergent P in sewage constituted approximately 30 percent (Natl. Swed. Environ. Prot. Board, 1972). The detergent-P per capita was reduced from 1.8 to 2 g/p/day to 0.9-1.1 g/p/day.

The experiences obtained by using NTA as a detergent chemical have not given the National Environment Protection Board any reason either to oppose this use or to work for a general changeover to NTA. The Protection Board considers advanced waste-water treatment for P-removal a safer and more effective method to reduce the phosphate load on our waters. The desire remains, however, for further limitation of the detergent phosphates.

The NTA-based or totally P-free detergents have never dominated the Swedish market. Therefore, it is difficult to evaluate possible environmental effects of these products. In any case, no harmful effects of NTA have been reported.

In Sweden as well as in most European countries, washing processes often are programmed at 80 to 90°C. Bleaching is then obtained by perborate, active above 60°C. Environmental aspects of boron have been summarized (R. Swed. Acad. Sci. 1970). Boron and its compounds (including perborates) have not been found to be acutely hazardous to the environment. The discussions concerning possible harm to the environment also included fluorescent whitening agents and enzymes. No special measures against these compounds have been taken. Fluorescent whitening agents were evaluated at the Stockholm Symposium in 1973, arranged by the Center for Environmental Sciences, Royal Institute of Technology, Stockholm (reported in MVC-Report 2, 1973).

To be able to follow changes in detergent formulations which might affect the environment, the detergent producers submit annual reports to the Environment Protection Board on detergent quantities and ingredients. They also provide information on the levels of phosphorus and organic chelators in each individual product.

CAMPAIGN FOR PHOSPHATE—FREE DETERGENTS IN NORWAY

Eutrophication problems exist mainly in southeast Norway. Recently, the largest lake in Norway, Lake Mjosa, with a surface of 365 km^2 showed very bad water quality due to a bloom of the blue-green alga *Oscillatoria bornetii* fa. *tenuis*. The alga discolored the water and gave the drinking water for 200,000 people very unpleasant taste and odor (Holtan, 1979; Holtan, et al. 1980). As phosphorus has been demonstrated to be the algal growth-limiting nutrient (see e.g., Ryding, 1980), a campaign aimed at reducing phosphorus pollution and saving this lake was initiated (Minist. Environ. 1979). Only a total removal of P was considered sufficient in the Mjosa area. This was thought possible because of the normally very low water hardness in the water supplies.

In the spring 1977, the environmental protection authorities gave new guidelines for sale of detergents in Mjosa's catchment area. The sale of phosphate-free detergents was to be emphasized. After 6 months phosphate-free detergents' share of the market rose from 2 to 3 percent to 57 percent. To increase the use of P-free detergents further, in February 1978, the Ministry of Environment issued regulations in pursuance of the Act concerning Product Control. These regulations "prohibit the exhibition of household detergents containing phosphates near the front of shop premises and also advertising of such detergents." The lack of a total prohibition derives from consideration for consumers using hard water. In early summer 1978 the turnover of P-free detergents had risen to about 70 percent. Decisions on whether taxes on detergents or other measures are required will be made.

The Mjosa campaign includes measures against pollution from municipalities, rural areas, agriculture, and industry at a cost of approximately 1 billion Norwegian crowns.

From 1973 through 1976 the annual load of total-P on Lake Mjosa averaged 320 tons/year. For 1977 through 1979 the corresponding values were between 230 to 252 (Holtan, et al. 1980). When the campaign is completed, discharges of P will be about 200 tons/year (Minist. Environ. 1979).

Since 1976 there has been no blue-green algal bloom. An evaluation of the measures performed will take some time as the period with reduced P-load also had cold and rainy summers.

Generally, an agreement between government and industry limits the P-content of fabric-washing products to a maximum 5.5 percent P. For machine dishwashing products there is no limitation. For the Mjosa catchment area consumers are advised to use as little as possible.

The government is considering whether NTA can be used in detergents in Norway. The problems discussed are the residual concentration of NTA which can appear in natural waters and in drinking water, and possibilities of heavy metal mobilization.

NO SPECIAL DETERGENT MODIFICATIONS IN DENMARK

As far as is known, no comprehensive detergent modifications resulting from environmental problems have been performed in Denmark. Detergent P has not been considered to be a problem in Denmark so far since sewage is normally discharged into the sea. In Finland an agreement within industry limits the P content of fabric-washing products to 7 percent, as P. NTA has been used as a substitute in Sweden.

DISCUSSION

The modified detergents have had no dominating position on the Swedish market, which means that it is difficult to evaluate their environmental influences. The rapid development of advanced wastewater treatment for P-removal eliminated the need for a more total replacement of P in the detergents. In rural areas, however, the detergent P, in addition to other P-sources, may still contribute to the over-fertilization.

In Norway, the measures taken to reduce the detergent P-consumption are difficult to evaluate, as other P-sources were reduced parallel to the decreasing sale of P-phosphoric detergents (Minist. Environ. 1979).

The NTA-containing detergent is still used in Sweden and Finland. As no large-scale conversion over to NTA has been attempted, this type of detergent has been used to a limited extent. In any case, no harmful effect of NTA has been reported. According to detergent experts, it is normally not possible to replace all sodium tripolyphosphate by NTA. The reason is that sodium tripolyphosphate has two different properties: (a) softens the water; (b) prevents precipitation — especially of calcium carbonate — during a normal washing and rinsing cycle. The effect of (a) can be achieved with, for example, NTA, citric acid, or other carboxylic acids.

The effect described in (b) is difficult to achieve with other substances, but requires only 5 to 10 percent sodium tripolyphosphate even in hard water.

If a phosphate-free detergent is required, all the components must be carefully chosen to prevent precipitation of Ca salts in hard water. If precipitation occurs, the visible washing result will be bad. This means severe restrictions on using different components and phosphate-free detergents have had no success.

Because of the very high chelating power of NTA the residual concentration of Ca ions is very low in the washing liquor when a NTA-based detergent is used. This gives a better washing power, especially on pigment dirt, than a normal phosphate-based detergent.

Totally P-free detergents have also been used. Some washing problems in hard waters were reported when these products were based on citrate (15 percent). Unfavorable costs for the substitutes compared with polyphosphates, increasing consumption of standard "low price" products, and the development of effective methods for P-removal in sewage, at present provide no base for successful marketing of modified detergents.

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THE LONG RANGE TRANSPORT OF AIR POLLUTION AND ACID RAIN FORMATION

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ABSTRACT

The increasing acidification of the precipitation in Europe was first pointed out in 1968 by Oden, who related this to the acidification observed in rivers and lakes in Scandinavia and the increasing use of fossil fuels with a high content of sulfur. In the OECD project "Long range transport of air pollutants" (1972/77), the acidification of the precipitation was quantitatively related to the emission and transformation of sulfur dioxide to sulfuric acid in the atmosphere. It was shown that extensive exchange of air pollutants took place between the European countries, and in orographic precipitation areas frequently exposed to polluted air masses, excessive amounts of acid precipitation were observed. Later studies have shown that the air pollutants from Europe also find their way into the Arctic region, particularly in the winter. The main acid component of the precipitation is sulfuric acid with an addition of 20 to 50 percent of nitrate and ammonium ions on an equivalent basis. The sulfate content is largely explained by the sulfate in the aerosol phase. The content of nitrate and ammonium ions is explained by the uptake of gaseous nitric acid and ammonia from the atmosphere. Atmospheric dispersion is discussed in relation to the methods used to describe the chemical transformations and the dry and wet deposition processes.

INTRODUCTION

The increasing acidification of the precipitation in Europe was first pointed out in 1968 by Oden. Data from the European Precipitation Chemistry Network coordinated by the Institute of Meteorology at the University of Stockholm, showed that a central area in Europe with highly acid precipitation (pH 3 to 4) had expanded to include also the southern part of Scandinavia.This observation was associated with observed acidification of the water in rivers and lakes in Scandinavia, where in many places the fish population had disappeared.

In addition, incidents of greyish snow were observed in areas remote from pollution sources. Chemical analyses of the polluted snow showed a high content of sulfuric acid, soot, fly ash, and other pollutants. These observations caused much alarm in Scandinavia, and in 1969 the matter was brought to the attention of the Organization of Economic Cooperation and Development. In the subsequent OECD project "Long range transport of air pollutants" (1972-1977) the acidification of the precipitation was quantitatively related to the emissions of sulfur dioxide in Europe (OECD, 1978; Ottar, 1978a). It was shown that extensive exchange of air pollutants took place among the European countries so that national control programs may achieve only limited improvements with respect to the total deposition of sulfur within the national borders.

Subsequent studies have shown that the air pollutants from Europe also find their way into the Arctic regions, particularly in the winter.

As a result of the OECD project, a European monitoring and evaluation program for the long-range transfer of air pollutants (EMEP) was established under the auspices of the U.N. Economic Commission for Europe and in cooperation with the U.N. Environmental Program and the World Meteorological Organization. Its main objective is to "provide governments with information on the deposition of air pollutants, as well as on the quantity and significance of long range transmission of pollutants and transboundary fluxes." At present, about 20 countries from both eastern and western Europe participate in the program; its design broadly follows that of the OECD program.

In North America the long-range transport of air pollutants is examined in several regional programs. While the European and Canadian studies centered on ecological problems resulting from the acidification of the precipitation, the U.S. emphasis was initially on air pollutant concentrations, health effects, and visibility. In later years this has changed, and the U.S. studies today also deal with acid precipitation problems. Most of the North American programs are described in the proceedings of the Dubrovnik symposium (Ottar, 1978b).

ACIDIFICATION OF THE PRECIPITATION

The general plan of the OECD project was simple. Single layer atmospheric dispersion models, wind trajectories, and an emission survey for sulfur dioxide were used to calculate the concentration fields of sulfur dioxide and sulfate on particles. The dry deposition was assumed to be proportional to the air concentration, and the annual deposition of sulfate by precipitation was empirically found to be proportional to the product sum of amount of precipitation and sulfate aerosol concentration. Parameters for the chemical transformation of sulfur dioxide to sulfate and deposition rates were adjusted by fitting the model to daily measurements from more than 70 ground stations in the region. Aircraft sampling was used to obtain information on the vertical distribution of sulfur.

The calculation was carried out in a grid system (a side length of 127 km) covering the northwestern part of Europe, and complete mixing was assumed up to a height of 1,000 m. Trajectories were calculated each 6 hours from wind fields obtained from the WMO Weather Service. With some improvements the same general approach is used in EMEP.

In Europe the spatial distribution of sulfur dioxide emissions follows the population density and location of major industries (see Figure 1) (Semb, 1979). The maximum concentration of sulfur dioxide is found near the major emissions. In the central part the annual mean concentration of sulfur dioxide is about 20 ug/m^3. Because Europe is situated in the westerlies, the maximum values are found slightly northeast of the emissions (see Figure 2) (Eliassen, 1978). The annual concentration pattern of sulfate particles is similar, but because of the time required for sulfur dioxide to be transformed into sulfate particles, the maximum concentration level is lower, about 10 μg/m^3. The dry deposition of sulfur dioxide is a significant factor in the central part of the area and responsible for removing about 50 percent of the total emission. Compared to this, the dry deposition of sulfate is of less significance. As shown in Figure 3, the annual deposition of sulfate by precipitation is strongly influenced by the amount of precipitation. Maximum deposition is found in orographic precipitation areas frequently exposed to polluted air masses. Examples are the Scandinavian mountains, the Alps, and mountains in Scotland. About 30 percent of the total emission is removed by precipitation.

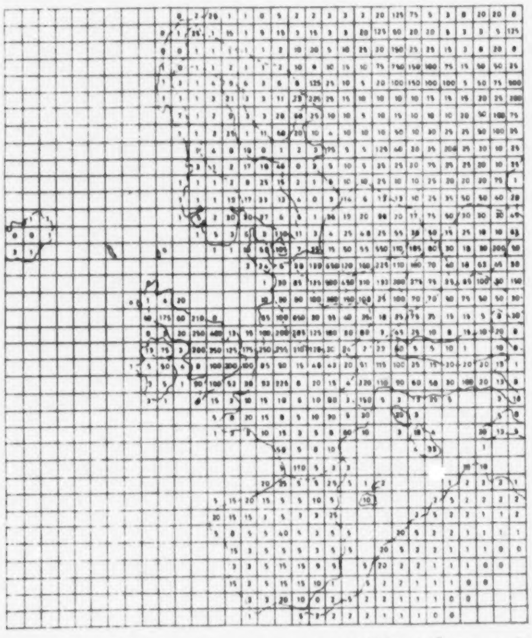

Figure 1. — Estimated annual emission of SO_2 (10^3 tonnes S) in grid elements with length 127 km at 60O N.

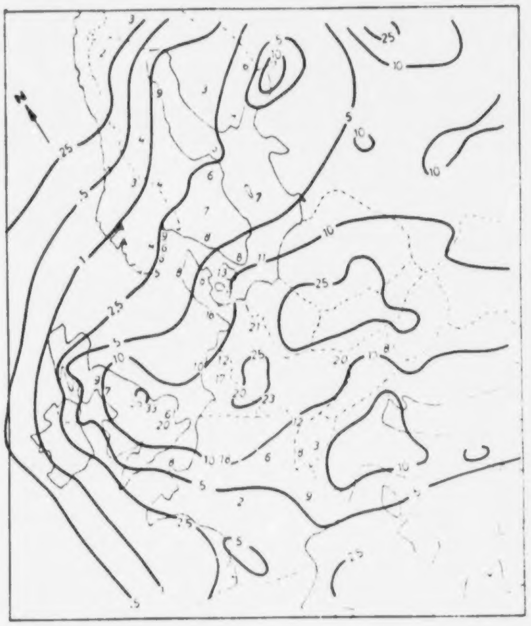

Figure 2. — Estimated mean concentration field for sulfur dioxide for 1974. Observed mean concentrations given by italic numbers Unit μg SO_2/m^3.

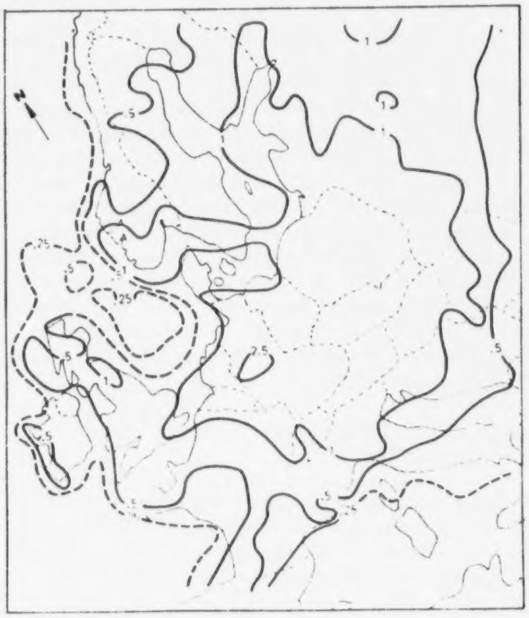

Figure 3. — Estimated sulfur wet deposition pattern for 1974. Unit: g S/mθ.

The day to day situation is very different from this average picture. With southerly winds, concentrations of 20 to 30 μg/m^3 of sulfur dioxide and sulfate particles are frequently observed in the Scandinavian area in places where such concentrations cannot be explained by local sources. About 50 percent of the total annual deposition of sulfate may result from about 10 episodes with highly acid precipitation. Aircraft measurements have shown that these polluted air masses lose little of their pollution content by passing over the North Sea, a distance of about 800 km. A similar situation is observed in other remote areas exposed to orographic precipitation. In 1978 an exceptional case of 10 mm precipitation with a pH of 2.5 was observed in Iceland. Precipitation with pH down to 2.4 is known from both Scotland and the west coast of Norway.

Recent studies have shown that in winter considerable amounts of air pollutants find their way from Europe and the Soviet Union into the Arctic (Larssen and Hanssen, 1979; Rahn and McCaffrey, 1980). Concentrations as high as 6 and 4 υg/m^3 of SO_2 and sulfate have been measured at Bear Island and Ny Alesund on Spitsbergen. These pollutants have been traced all the way across the Polar Basin to Barrow in Alaska. There is very little precipitation in this region during the winter, and evidently the chemical transformation rate of sulfur dioxide is much reduced.

The main acid component of the precipitation is sulfuric acid with an addition of 20 to 50 percent nitrate and ammonium ions on an equivalent basis. The sulfate content of the precipitation is largely explained by nucleation on ammonium sulfate and ammonium hydrogen sulfate from the aerosol phase. The content of nitrate is probably explained by the absorbtion of nitrogen dioxide and gaseous nitric acid from the atmosphere.

In Scandinavia the concentration of sulfate in precipitation is generally highest during the spring, while the emissions of sulfur dioxide in Europe reach a maximum in January (about twice the emissions in July-August). This delayed maximum sulfate concentration in precipitation can be attributed to a precipitation minimum in western Europe during the early spring, and more rapid conversion of sulfur dioxide to sulfate with increased solar radiation (Joranger, Schaug, and Semb, 1980). The seasonal variation of the concentration of nitrate in precipitation is similar but with a longer maximum period. For further elucidation of these differences, comparison should be made between air and precipitation concentrations of nitrogen compounds as well as for the sulfur compounds.

MODELING OF THE LONG RANGE TRANSPORT

Our knowledge of the details of long-range transport of air pollutants and the acidification of the precipitation is limited by the methods used. The following discusses the significance of some of these limitations.

Emissions

The sulfur dioxide emissions in Europe are due mainly to the burning of sulfur-containing coal and oil. The increased demands for energy after 1950 were met by widespread introduction of petroleum products, and as a result sulfur dioxide emissions in Europe doubled from 1950 to 1970 (see Figure 4) (Semb, 1978).

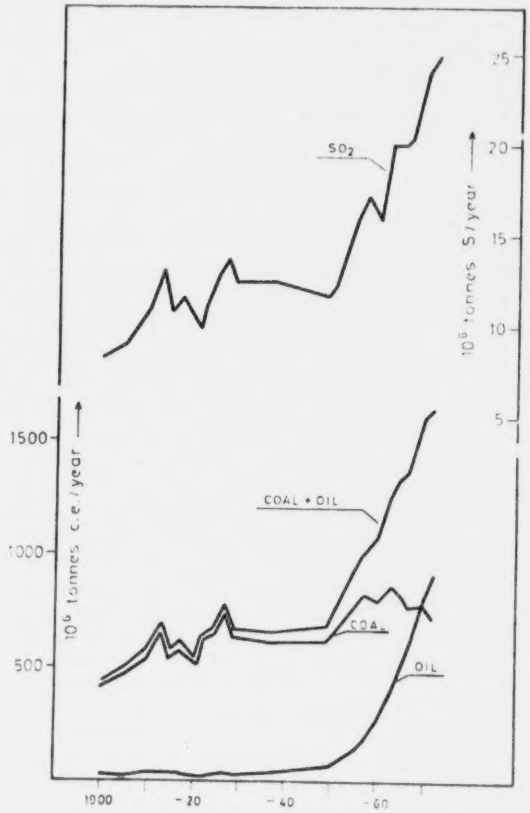

Figure 4. — Fossil fuel consumption and estimated sulfur dioxide emissions in Europe.

When this emissions increase is considered in relation to the transport of the air pollutants to remote areas, it may well be that polluted precipitation has occurred for a long time without being noticed. Thus, fish kills in rivers in southern Norway reported at the beginning of this century may well have resulted from long-range transport of sulfur pollutants. The decline in fish populations has been much more dramatic in the last 30 years, however.

The emission survey for the OECD study was established in coooperation with the participating countries. For other countries this survey was based on national fuel consumption data collected from OECD and ECE, emission factors, and population density. For some countries the accuracy is at least within 10 to 15 percent. This survey is being further elaborated in connection with EMEP.

The size of the grid element limits the geographic resolution, and the atmospheric dispersion models can give only a smoothed picture of the concentration fields. Clearly, measurements used to verify the model calculations should represent comparatively large areas. Furthermore, the acidity of the precipitation also depends on other chemical components present, particularly the nitrate and ammonium ions. It is therefore also necessary to know the emissions of nitrogen oxides and ammonia. Detailed emission surveys for nitrogen oxides have been constructed for some European countries (Semb, 1979). Recently, Bonis, Meszaros, and Pusey (1980) estimated the nitrogen oxide and ammonia emissions for all Europe.

It may be asked how relatively inaccurate emission surveys can yield useful information. The answer is that, however uncertain, emission surveys are an indispensable tool in understanding the occurrence and dispersion of air pollutants. The accuracy in general should be ±20 percent or better, and the positions and relative emission strength of major emission areas are reasonably well defined. Although better data would be welcome, the accuracy of the present survey is sufficient for dispersion model calculations.

Transport

Aircraft measurements show that the air pollutants usually remain below a mixing height of 1 to 2 km, and 100 to 200 km downwind of a source area; there is no further rapid dilution of the pollutants. Beyond this distance, which depends on the weather conditions, the pollutants are slowly removed by dry deposition. The only process which can rapidly clean the air is precipitation.

When air pollutants have reached this state of dilution, the transport of the polluted air masses is conveniently described by wind trajectories in a grid system. In the grid models calculations are based on average values for each grid element with respect to emissions, wind, rainfall, etc. The geographic resolution of these models is limited by the size of the grid element. There is also a relation between the geographical and the time resolution which can be obtained. For instance, the effect of nocturnal ground inversions cannot be described in a simple one-layer dispersion model. Therefore, 24 hourly measurements will fit the model better than 6 hourly measurements. To include such variations one has to use a smaller grid element, a two-level model, or a perturbation of the vertical concentration profile within the grid element. The main problem is the effort required to provide measurements to verify the results of more detailed calculations.

Two different types of models were used in the OECD project (Eliassen, 1978). In the back trajectory model, the uptake and deposition of air pollution is calculated for an air parcel following the trajectory up to the point of interest. In the OECD program the concentration for each grid element was calculated from 48-hour back trajectories, and compared with measured daily mean concentrations. In the EMEP, 96-hour back trajectories are used to reduce the amount of pollution of unknown origin.

In this model the contributions to one grid element from all other elements are easily separated, and the model is regularly used to calculate the exchange of pollution between the European countries. In the Lagrangian model of the OECD project, forward trajectories were used to calculate the concentration field with regular time intervals. This model has an unlimited memory and can be used to predict episodes of air pollution using weather forecast data.

In both models the air parcel is assumed to follow a calculated trajectory. However, this trajectory does not represent a physical reality, as the lateral and vertical dispersions are neglected. The small scale turbulence is not significant, but the meso-scale wind variations cannot be neglected. These are simulated in an indirect way in the two models mentioned by the fact that the concentration values represent averages for large grid elements. This introduces a so-called psuedo-diffusion, the magnitude of which is determined by the size of the grid element, the time step used in the calculation, and the numerical advection procedure.

Husar and Patterson (1979) have recently developed a different model based on individual handling of a stream of air pollution parcels from each emission source. The source strength is given by the number of parcels and not by the concentration of each parcel. To account for the meso-scale dispersion, they have introduced a random displacement of the air parcels when they have passed along the calculated trajectory for a specified time interval. Probability distribution functions are used to account for chemical transformations and deposition probability.

A main advantage of this model is that the lateral (and if necessary the vertical) dispersion is separated from the choice of grid size. For models on a global scale, this may be an essential feature. A serious limitation of this model in its present state is the requirement of linear chemical interactions. For sulfur dioxide and the formation of sulfates this causes no problem, but in the case of nitrogen oxides and nitrates chemical reactions are far from linear; this raises the important question of using simplified procedures.

Modeling the long range transport of air pollutants involves a number of approximations, some of which have been mentioned. Because of this, the day to day agreement between observed and calculated concentrations is reduced. For mean values over extended periods of time better agreement is usually obtained. Principally, the same applies to mean values for larger areas, but most of the measurements represent point values, normally at ground level. In principle, mean values for larger areas, perhaps observed from satellites or aircraft, should give better agreement.

Chemical transformation and deposition rates

In calculating the long-range transport of air pollutants constant transformation and deposition rates are generally used, and the wet deposition is often estimated from annual precipitation data. In the OECD project wind fields at different levels and a number of advection schemes were tried, but a sensitivity analysis showed that it would be more important to improve the modeling of the chemical transformation and deposition. As a first step, daily precipitation fields are estimated and used to calculate the wet deposition in EMEP. In this case the necessary data are available from the WMO Weather Service.

Available information on the dry deposition rate of gases and aerosols (Garland, 1978) is generally limited to results of special laboratory and field investigations. Although there is considerable evidence of variations in the dry deposition velocities for different surfaces, seasons, and weather conditions, constant deposition rates are generally used for all seasons and surface areas in the long-range transport models.

This is not a satisfactory approach, particularly when transport over very long distances is considered. In the summer season the sea is colder than the air, and a shallow, stable layer of air often forms over the sea surface, reducing vertical mixing of the air. In winter the North Sea and the Atlantic are generally warmer than the air, leading to increased vertical mixing and precipitation, while the continental land masses and frozen water bodies are colder than the air, forming stable stratification near the surface.

As a first approximation one might correct for these effects by introducing different deposition rates for summer and winter and for land and sea areas. However, to justify this additional information is required. A simple calibration of the model is highly unsatisfactory.

Similar conditions apply to the chemical transformation rates. Studies in the Arctic region and statistical analyses of data from the OECD project (Prahm, et al. 1979) strongly indicate that the transformation rate of sulfur dioxide to sulfate decreases with concentration and depends on temperature, sunlight, and the presence of other pollutants. Again, more measurements are needed to specify these conditions in the dispersion models.

The oxidation of sulfur dioxide to sulfate follows two main pathways. When sulfur dioxide is absorbed and oxidized catalytically in cloud droplets, the absorbtion stops if the droplets become too acid. Ammonia will neutralize the acidity and thus make further absorbtion possible. Over the sea, little or no ammonia is available, and the reactions stop. Photochemical oxidation in the gaseous phase then becomes relatively more important, and this leads to the direct formation of sulfuric acid droplets. Under these circumstances pH-values down to 2.5 have been observed in coastal precipitation.

The catalytic oxidation of sulfur dioxide is much more rapid in plumes from coal combustion than from oil, because of the manganese content in submicron fly ash particles from coal. Thus, the transformation of sulfur dioxide to sulfate may go faster when polluted air from the European continent passes over the Scandinavian area. The gas phase oxidation of sulfur dioxide is intimately related to the photochemical reactions of the nitrogen oxides and the production of hydroxyl ions.

A normal rain shower in Scandinavia precipitates approximately 1 ml of water from each m^3 of air at the level where the precipitation is formed. Comparisons of the sulfate content in precipitation with the aerosol sulfate concentration at ground level show that the amount of sulfate in precipitation corresponds to complete scavenging of the sulfate particles at the level of rain formation with only a minor addition of sulfate from the absorption of sulfur dioxide.

On the other hand, the experience that 20 to 50 percent of the acidity in precipitation may be from nitric acid, while simultaneous measurements of particles show little or no nitrate ions, is evidence that most of the nitric acid in precipitation does not come from the particles, but probably from gaseous nitric acid. More measurements of gaseous nitric acid, ammonia, and the composition of cloud droplets and aerosols are needed to clarify the significance of these processes.

The sulfate particles responsible for acidifying the precipitation are found in the accumulation phase of the bimodal aerosol size distribution (0.1 to 2.5 μm). The sea salt particles are mainly found in the larger fraction (above 2.5 μm). Samples collected at coastal stations therefore are corrected for their content of sea salt sulfate by analyzing for sodium, chloride, or magnesium.

The small particles in the accumulation mode are important in the long-range transport of air pollutants, and their chemical composition is markedly different from the larger particles. Size-segregated sampling would prevent chemical reactions between the small and the large particles on the filter, which for instance mav result in a loss of hydrochloric or nitric acid, and thus assist in interpreting the results.

CONCLUSIONS

The studies of the long-range transport of the atmospheric sulfur pollutants that began in the 1970's, have shown that the air pollutants are more widely distributed than previously believed. The components that are transported over long distances as gases and as particles in the accumulation mode, include most of the pollutants and their secondary products.

The acidity of the precipitation is governed mainly by its content of sulfate, nitrate, and ammonium ions, and may to a large extent depend on the pathway of the polluted air masses. The lowest pH values are obtained in air masses which have remained over the sea for a longer period of time.

The modeling of the long-range transport and the formation of acid precipitation includes many simplifications. For larger areas and longer periods of time the agreement between observed and calculated values is reasonably good. To improve the day to day agreement, the chemical transformations taking place in the atmosphere and the deposition processes have to be described in more detail.

The photochemical oxidation of sulfur dioxide is intimately connected with the photochemical reactions of the nitrogen oxides; however, introduction of nonlinear chemical reactions in the atmospheric chemistry of the dispersion models will seriously complicate the models. Adequate simplified procedures must be developed.

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EFFECTS OF ACID PRECIPITATION ON AQUATIC AND TERRESTRIAL ECOSYSTEMS

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ABSTRACT

Acid precipitation, characterized by high concentrations of H^+, SO_4 and NO_3, occurs over large regions, notably in Europe and eastern North America. In areas susceptible to acidification, i.e., areas with sparse soil cover and bedrock geology poor in neutralizing minerals, acid precipitation acidifies waters. The normal bicarbonate buffering system breaks down, and sulfate becomes the dominant anion in acidified water sources. Increased inflow of aluminum from the soil to the lakes is particularly important to aquatic life. Aquatic ecosystems in acidified areas often show a simplified structure, where a few tolerant species dominate. Changes are seen on all trophic levels. Populations of valuable fish species, especially salmonids, are reduced or wiped out in many acidified districts. There are signs that nutrients like calcium and magnesium have been reduced in some soils exposed to acid precipitation. Continued leaching may eventually have negative effects on forest growth. At present, field evidence of reduced growth is inconclusive, and experimental research has in some cases shown growth increase under acid conditions. This is interpreted as a fertilizing effect of the nitrogen content in acid precipitation.

EXTENT OF FRESHWATER ACIDIFICATION

Acidification of lakes and rivers during recent decades is a regional problem in Scandinavia and eastern North America. The acidified areas are underlain mainly by siliceous (quartz-rich) bedrock with sparse or thin soil cover. These same areas now receive decidedly acidic precipitation (weighted average below pH 4.6), and the time trends in acidification of precipitation and inland waters are parallel. Recent acidification of freshwaters is normally not found in geologically similar, sensitive areas which lie outside the regions of acid precipitation (e.g., Likens, et al. 1979; Wright, et al. 1980).

This regional coincidence both in space and time strongly suggests that aquatic ecosystems are being acidified by atmospheric deposits. The extent of acidification is known from a few existing observations of water pH and other chemical characteristics over the years, and indirectly through mapping of lakes and rivers where fish populations have been reduced or lost in recent years. Nothing but acidification with its associated altered chemical conditions can explain the present regional fish loss. Surveys of land use changes associated with agriculture and forestry practices in acidified parts of Norway show no systematic relations with acidified lakes and fish population loss (Drablos, et al. 1980).

Observations from 1920 to 1970 of pH in 128 lakes in southern Norway have been compared to pH data from the same lakes during the 1970's. Of these lakes, 63 percent had become at least 0.25 pH units *more* acid and 12 percent had become at least 0.25 pH units *less* acid. Only 4 percent of the lakes had pH below 5.0 prior to 1950, compared to 25 percent in 1977. Before 1950 none of 130 lakes in southern Sweden was below pH 5.5. In 1977 28 percent were below pH 5.5 and 15 percent below 5.0. All the lakes that had become more acidic are situated in areas of southern Scandinavia that today receive acid precipitation with a pH below 4.6 (Wright, 1977).

Similar observations have been made in North America. Of 320 high elevation lakes in the Adirondack Mountains of New York, about 70 percent had a pH above 6.5 and only 4 percent below 5.0 in the 1929-1937 period. In 1975, 51 percent of a group of 217 high elevation lakes had pH below 5.0 and 90 percent of these lakes were devoid of fish (Schofield, 1976).

SOIL PROCESSES AND WATER ACIDIFICATION

Several processes are known to acidify soil:

1. Root uptake of cations during plant growth.
2. Carbonic acid formation from CO_2 derived from respiration of soil fauna and flora.
3. Oxidation of nitrogen and sulfur compounds to nitric acid and sulfuric acid.
4. Organic acids produced during decomposition of plant matter.
5. Atmospheric input of acidifying substances, notably sulfuric acid.

Close to emission sources the acidity produced as a consequence of dry deposition of SO_2 may dominate over acidity produced by precipitation.

When soils acidify, the most important effects probably are increased mobility and leaching losses of basic metal cations such as Ca^{+2}, Mg^{+2}, K^+, and Al^{3+}.

The ion exchange processes in soil are very important for soil and water acidification. Soil particle surfaces are normally negatively charged, and therefore surrounded by cations.

The cations, including hydrogen ions, can be exchanged between the soil particles and the soil water solution percolating in the soil pores. What cations are exchanged depends on their charge and other properties, and on the relative amounts in solution and adsorbed to the soil. At high concentrations, hydrogen ions in the percolating water will tend to exchange with calcium, magnesium, and aluminum ions. This results in higher concentrations of Ca, Mg, and Al in the soil water, and lower H^+ concentration, which means a neutralizing effect on the soil water which eventually enters lakes and streams. Net adsorption of H also leads to a more acid soil, unless the H^+ ions are consumed in weathering processes.

Acidification of soil is a slow process in nature, and field detection of effects of additional inputs of acidifying components is likely to be difficult. There is agreement that sandy, well-drained soils of intermediate pH are particularly susceptible to pH changes. There are, however, very few indications from field studies of soil acidification caused by atmospheric deposition (Troedsson, 1980; Linzon and Temple, 1980).

As the amount of basic cations in a particular soil profile is reduced, a smaller proportion of H^+ will be adsorbed, and a greater proportion of inflowing hydrogen made available for transport to the watercourses.

The transport of cations from the soil to the water systems depends on available anions to maintain electrical charge neutrality. Sulfate ions are important as vehicles for cation transport as in many soils they are very mobile, and will be adsorbed only temporarily (Cronan, et al. 1978; Johnson, 1980).

Sulfate therefore plays a decisive role in the acidification of freshwater, as a mobile carrier of the hydrogen ions whether the hydrogen ions stem from atmospheric deposits or are produced in the catchment. The input and output of sulfate to catchments are in many cases close to balance over periods of several years. There is, however, often a retention in the winter snowpack and during dry summers, and releases during spring snowmelt and autumn rains (Likens, et al. 1977, 1980; Seip, 1980; Figure 1). These processes may produce episodes of very acid stream water, as the sulfate is washed out with equivalent amounts of cations, which in the acidified regions will tend to be hydrogen ions. To explain water acidification, it is therefore probably more important to consider the possibility for leaching of H^+ and other cations provided by sulfate, than the total amount of H^+ in the catchment (Seip, 1980).

The relationship between hydrogen and sulfate ions is demonstrated by data from regional surveys of Norwegian headwater lakes in 1975-1978. Lakewater chemistry was significantly correlated to precipitation chemistry. Sixty to 80 percent of the variance in lakewater content of H^+ and SO_4 could be explained by precipitation amount and content of H^+ and excess sulfate (Mohn, et al. 1980).

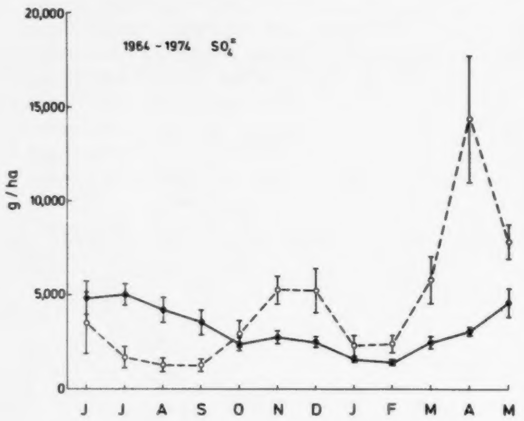

Figure 1. — Monthly flux of sulfate for undisturbed ecosystems of the Hubbard Brook Experimental Forest, N.H., showing input (solid line) dominance during summer, and output (dashed line) dominance during autumn and spring. (Likens, et al. 1977.)

LAKE ACIDIFICATION

The acidification process of lakes exposed to acid water inflow can be described as a large-scale titration (Henriksen, 1979, 1980). Weathering of rock material in the catchment provides bicarbonate, HCO_3, which normally is the major anion in soft-water lakes, with calcium, Ca, and magnesium, Mg, as the major cations.

Lakes with high bicarbonate levels are well buffered (i.e., they resist changes in pH levels) and have pH above 5.5. Fish populations are usually normal. High influx of strong acids, notably sulfuric, from the atmosphere may deplete the bicarbonate buffer and cause severe pH fluctuations resulting in physiological stress, reproductive failure, and episodic kills of fish.

If the influx of acids is high enough to completely exhaust the bicarbonate buffer, the lake will enter the acidified stage characterized by pH well below 5.0, sulfate instead of bicarbonate as the dominant anion, and high concentrations of aluminum, Al. Fish stocks are severely reduced or lost.

Calcium, which normally accompanies bicarbonate, is a useful indicator of the geological influence from the catchment upon water chemistry (Figure 2). Calcareous rocks and soils are easily soluble and will produce lake waters with high concentrations both of calcium bicarbonate and other compounds which provide buffering capacity.

Thus waters in areas with calcareous bedrock and soils, such as much of central Europe, will not be acidified in spite of the fact that the acidity of precipitation is very high. However, when a major emission source happens to be located close to geologically susceptible areas, such as the metal smelters at Sudbury (Ontario), Canada, which have annual emissions near 1.35 million tons SO_2, the chemical and ecological effects on the environment can be devastating.

Particularly high concentrations of hydrogen ions and other substances are commonly observed during early spring snowmelt. Both laboratory experiments and field observations have shown that concentrations can be 3 to 10 times higher in the first meltwater than in the bulk snowpack (Johannessen and Henriksen, 1979). Although there is considerable contact between meltwater and soil (Seip, 1980) modifying the chemical properties, the snowmelt period often produces major impacts on aquatic chemistry and biota.

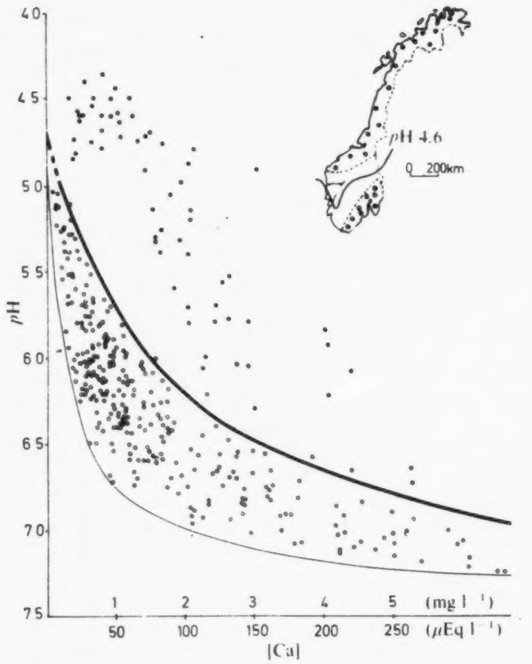

Figure 2. — pH and calcium concentrations in Norwegian lakes 1974-1977. Lakes in southeastern Norway (●) receive highly acid precipitation, pH 4.2 — 4.5.

When calcium concentrations are assumed to be well correlated with pre-acidification bicarbonate alkalinity, the empirically drawn curve will distinguish between acidified and nonacidified waters. (Henriksen, 1979.)

EFFECTS OF ACID WATER ON AQUATIC LIFE

The recent acidification of freshwater in parts of Europe and eastern North America has had profound impacts on aquatic life. All trophic levels have been affected. The most immediate concern to the people living in the acidified regions is the major decline in fish populations, but primary producers, decomposers, and invertebrate animals also are affected. (Almer, et al. 1978).

Reduced numbers of several algal species have been observed in acid lakes, especially among green algae. On the other hand, there is often a conspicuous heavy growth of filamentous algae and mosses in many acid lakes and streams. The algal accumulation is probably caused by reduced activity of invertebrates feeding on the vegetation, and reduced decomposition. The dominance of a few plant plankton species in acid water probably results from specific tolerance or changed biological interactions. Many of the algae are, however, photosynthetically inactive, and thus the productivity per unit of biomass may be lower in acid waters. A possible factor reducing productivity in lakes of pH 5 to 6 is precipitation of phosphorus by aluminum released to the lakes from the surrounding catchment (Almer, et al. 1978).

Expansion of sphagnum moss on bottoms of acidified lakes is known from Sweden (Grahn, et al. 1974); sphagnum mats are also reported from south Norway (Hendrey, et al. 1976) and the acidic Lake Colden in the Adirondack Mountains of New York (Hendrey and Vertucci, 1980).

The silica-containing algae, known as diatoms, show changes in community composition, shifting to more acid-tolerant species in rivers and lakes under acidification.

Diatom remains in sediments in south Norwegian lakes indicate that lake water pH has declined 0.5 pH units or more since about 1930 to 1945 (Davis and Berge, 1980).

Among decomposing organisms in acidified lakes there is a shift from bacteria to slow-acting fungi, leading to increased accumulation of organic matter and reduced availability of nutrients. This is observed both in North America and Scandinavia.

The invertebrate fauna is an important link between primary producers and fish in the aquatic food chain. Both zooplankton, aquatic insects, non-planktonic crustaceans, snails, and mussels are reduced in abundance and diversity during water acidification. A few examples from Norway may illustrate:

Norwegian studies of mayflies indicate that the mean number of species is about three to four times higher in water with pH 6.5 to 7.0 than at 4.0 to 4.5 (Leivestad, et al. 1976).

The mayfly *Baetis rhodani* is usually a key organism in the food chain in oligotrophic rivers, transferring energy from plants to the higher stages. This species comprises 60 to 80 percent of the mayflies or even more in parts of Norway. The species occurs in less acid rivers, ph > 6.0, all over the country, and produces one to two generations per year. In water of 4.5 to 4.7 and low salinity ($\kappa_{20} = 30 - 35\ \mu S/cm$), *B. rhodani* cannot survive more than 2 days. At the same pH, but higher salinity ($\kappa_{20} = 125 - 130\ \mu S/cm$), 10 percent of the animals were still alive after 5 days. Field observations indicate that *B. rhodani* fails to reproduce and dies from physiological stress in water of ph < 5.0 (Raddum, 1979).

The freshwater shrimp, *Gammarus lacustris*, is one of the most important food organisms for trout in Norway. In one oligotrophic lake studied it constitutes 24 percent of the energy intake of trout (Lien, 1978). In lowland lakes it has not been recorded below pH 6.6. Experimental tolerance tests have shown that adult *G. lacustris* can be eliminated during short-term acidification below pH 5.5 (Hendrey, et al. 1976).

Freshwater snails, bearing calcareous shells, are generally not found below pH 6.0. Only 5 of the 27 Norwegian species occur in lakes between pH 6.0 and pH 5.2. Also the abundance of snails is reduced in acid water. Small mussels also disappear around pH 6.0. Only 3 of the 20 Norwegian species have been found in lakes below pH 5.0 (Okland and Okland, 1980).

EFFECTS OF ACID WATER ON FISH

A regional decline in inland fisheries during the last decades has been reported from acidified districts of south Scandinavia, Canada, and the United States (Muniz and Leivestad, 1980; Harvey, 1980; Schofield, 1976).

Fish decline in Canada was first reported in the 1960's from the La Cloche Mountains near Sudbury. In this region 33 of 150 lakes were classified as "critically acidic" with pH below 4.5 and 37 more lakes as "endangered," pH 4.5 to 5.5.

Today, perhaps 200 lakes in Ontario are known to be devoid of fish, because of acidification. In Nova Scotia a dozen salmon rivers now show pH's in the 4.5 to 5.0 range, and the salmon catch is declining (Harvey, 1980).

Intensive studies of acid precipitation effects on fish populations in 217 lakes in the Adirondack Mountains showed that in 1975, more than half of the lakes had pH below 5.0 and 90 percent of these lakes were devoid of fish. Comparable data from 1929-1937 indicated that only 4 percent of these lakes were below 5 pH and devoid of fish. Entire fish communities (brook trout, lake trout, white sucker, and others) were eliminated over a period of 40 years, resulting from decreased pH (Schofield, 1976).

In south Scandinavia the first effect of acidification on fish became known early in this century when salmon began to disappear from several southern rivers, all of which are now acidic. Some of these rivers have now lost their salmon completely. In Sweden, the roach disappeared from some west coast lakes as early as the 1920's and 1930's. It is estimated that in the Swedish west coast region, which is most sensitive, 50 percent of the lakes now have pH below 6.0. For the whole of Sweden, the number of lakes with pH below 6.0 is now about 10,000 (Dickson, 1975). Also fish populations of char, perch, and pike have been seriously affected. In south Norway, fish population surveys of more than 5,000 lakes have shown that within an area of 13,000 km^2 fish life is now virtually extinct. In an additional area of 20,000 km^2 the lakes are losing their fish.

Some lakes are already barren, many have sparse and declining populations, and some still give reasonable fish yields. The population status since about 1940 is known for almost 3,000 lakes in the affected districts in southernmost Norway. There is evidence that the fish decline was moderate before 1940 and most pronounced since 1960. The number of remaining trout populations is quickly being reduced. At the present rate, the four southernmost counties of Norway will have lost 80 percent of their trout populations by 1990 (Sevaldrud, et al. 1980; Figure 3).

Regional data show a close correlation between increased water acidity and loss of fish. In lakes with low salt content the fish loss is greater than in lakes of higher salt content and the same pH level. It is also typical that small lakes at high altitudes lost their fish populations first. Today about 80 percent of lakes above 1,000 m above sea level are empty. The fish loss has since gradually spread downstream.

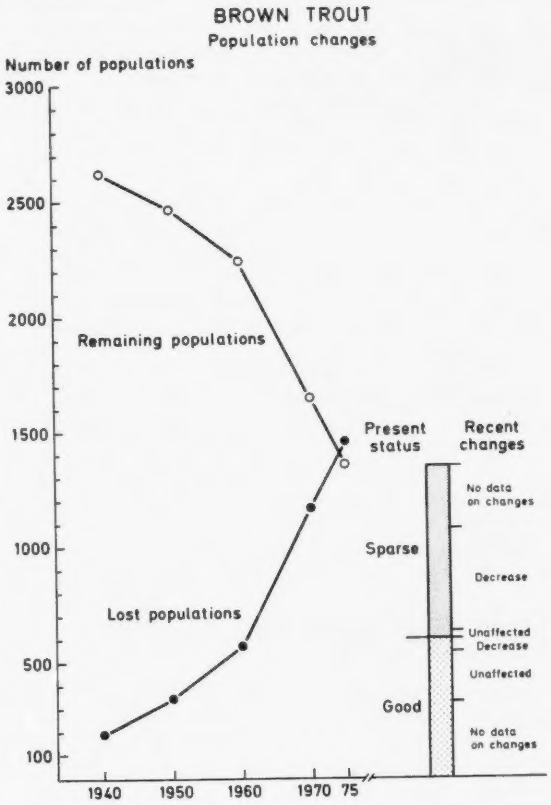

Figure 3. — Time trend for population losses of brown trout from the four southernmost counties of Norway. (Muniz and Leivestad, 1980.)

High egg and fry mortality in acid water that reduces younger age classes, is regarded as a main reason for fish decline (Schofield, 1976), but other population responses such as post-spawning extinction are also known (Muniz and Leivestad, 1980). Massive fish kills of adult fish during acid episodes, especially during snowmelt, are well documented.

Seemingly contradictory results from field observations and laboratory tests for fish survival in artificially acidified tap water have indicated that some toxic agent other than acidity increases mortality under field conditions. Aluminum which is present in high concentrations in lakes in acidified districts, is now held to be a critical element for fish mortality.

Exposure tests, field experiments, and physiological research have in recent years led to the following hypothesis for fish loss in acidified districts:

1. Aluminum is dissolved and leached from the soils of catchments receiving acid precipitation, and Al ions occur in high concentrations in acidified lakes and streams.

2. In acid, clearwater lakes low in organic content, the aluminum will mainly be present as inorganic compounds, some of which are highly toxic to fish, and probably to other aquatic animals as well.

3. The toxicity of Al in water varies with pH, having a maximum around pH 5. Aluminum toxicity thus acts in combination with the "pure" pH stress on fish physiology.

4. Aluminum toxicity attacks the gills. Al content in gills of acid-stressed fish may be six to seven times higher than in reference fish. This leads to mucus clogging. Aluminum disturbs the exchange of ions across the gill membranes. High concentrations of dissolved salts tend to ameliorate aluminum stress and ion depletion.

5. The main physiological effects of pH/Al stress are: (a) depletion of body salt content; (b) hyperventilation; and (c) lowered blood oxygen tension. Other effects are also observed.

6. In some species, like brown trout, metabolic activity increases, possibly reducing energy available for growth.

Salmonid fishes are generally more vulnerable to acid than other important species.

EFFECTS OF ACID PRECIPITATION ON VEGETATION

Anthropogenic sulfur may affect soil and vegetation mainly by two pathways: Sulfur dioxide is a primary air pollutant, acting directly on soil and vegetation growing close to the emission sources. Acid precipitation contains high concentrations of sulfate which is derived from sulfur dioxide. Acid precipitation has a much wider distribution than SO_2 and may indirectly affect vegetation through chemical or biological changes in the soil (Dochinger and Seliga, 1977; Abrahamsen, et al. 1976; Hutchinson and Havas, 1980).

The increase in anthropogenic sulfur emissions, coupled with the increased height of emissions, have led to the transport of sulfur pollutants over long distances. Acid precipitation from the polluted air masses may affect vegetation directly or indirectly by interfering with important soil processes.

The direct contact between acid precipitation and vegetation increases the leaching of some elements from the foliage. Precipitation also washes off substances dry-deposited on the vegetation. The total effect is an increase in concentrations of most of the compounds in throughfall compared to incident precipitation.

Leaching from foliage is high in cations such as calcium and potassium. There are indications of a pH-dependent loss, possibly as a result of exchange with H^+ ions. Leaching can lead to the appearance of deficiency symptoms in leaves. On the other hand, vegetation acts as an efficient filter of the chemical components in air and precipitation, and the cycling from litter-fall to the soil to root uptake can be intense, especially for plant nutrients such as nitrogen that are in high demand. The levels of lead found in organic matter in forest soils in remote areas in New England were comparable with those in many heavily traveled roadsides, and levels were rising (Reiners, et al. 1975).

Analyses of more than 500 moss and soil samples in Norway (Hanssen, et al. 1980; Allen and Steinnes, 1980) show that long-range transport of trace elements determines the distribution of lead, zinc, and cadmium, and to some degree arsenic, antimony, and selenium.

Direct effects of acid precipitation on forest trees have been shown experimentally. The wax coating of the outer layer (cuticula) of oak was eroded at precipitation pH 3.2, possibly affecting water loss and attacks by fungi and bacteria (Schriner, 1976).

Direct effects of acid precipitation on agricultural crops depend on the particular cultivar, on precipitation characteristics, and the growing conditions. Precipitation at pH above 4 seems to present a low risk of measurable reductions in growth or yield. At pH levels between 4 and 3 many effects on crops have been demonstrated, both positive and negative, and pH below 3 seems to substantially increase the chances of harmful effects on growth or yield (Jacobson, 1980).

PLANT NUTRIENTS AND FOREST GROWTH

Loss of nutrient minerals, a natural process caused by weathering, appears to be widespread and enhanced from soils in areas with high deposits of acidifying components. This has been observed in input-output balances for the Hubbard Brook catchment in New Hampshire (Likens, et al. 1977), from studies of nine Norwegian catchments (Wright, et al. 1978), in the Solling forest, the Federal Republic of Germany (FRG) (Ulrich, 1980), and in lysimeter experiments (Abrahamsen, 1979). The catchments generally act as temporary sinks for H^+, NO_3, and NH_4, and as sources for Ca, Mg, Mn, and Al. Sulfate is generally close to balance, but some studies (Uhrich, 1980; Andersson-Calles and Eriksson, 1979) indicate an accumulation over several years in catchments. This buildup is probably a recent process which started with large scale emissions of SO_2 from fossil fuel burning. If the deposition reaches a new and stable level, input and output are expected to balance once more after some time. Little is known of possible reemission of sulfur in gaseous form after deposition.

Recently published data from the National Forest Survey in Sweden illustrate the situation in that country (Troedsson, 1980). Chemical data from 2,500 humus layer sites in the forested area between 59° and 61°N show significant decreases in exchangeable calcium, magnesium, and potassium between 1961-1963 and 1971-1973. Exchangeable H^+ and aluminum have increased, but not significantly. The loss of Ca, Mg, and K from the soil is interpreted partly as an effect of atmospheric acid deposition. There is a strong correlation between increasing age of the coniferous forest and decreasing pH in the humus layer, and this effect is stronger than the acidifying effect resulting from atmospheric deposition (Troedsson, 1980).

When plant nutrients leach from soils exposed to acid deposition faster than minerals weather (which provides new dissolved compounds while consuming H^+ ions), the net loss may be important for plant productivity. Loss of magnesium due to soil acidification is already believed to restrict forest growth in parts of central Europe (Ulrich, 1980). Concentrations of aluminum in the soil solution are so high in some acid-impacted soils that Al may possibly be toxic to tree growth (Ulrich, Mayer, and Khanna, 1979; Voigt, 1980).

In soils with nitrogen and sulfur deficiency, acid precipitation could have a positive growth effect. Douglas fir stands in the Pacific Northwest region of the United States are only one example. At the same time it is suspected that acid precipitation may have depleted potassium in some soils (Johnson, cited by Roberts, 1980).

Effects of acid precipitation on forest growth are therefore now considered a nutritional problem (apart from possible direct effects). The increased deposition of nitrogen and sulfur can be regarded as fertilization, and the increased leaching of nutrient cations caused by increased atmospheric deposition of sulfur compounds will tend to cause nutrient deficiencies. Plant requirements for different nutrients and soil properties will determine whether the growth effects will be negative or positive (Abrahamsen, 1980).

Experiments on the effect of artificial acidification on forest growth under field conditions have been carried out in Sweden and Norway. The Swedish experiments have shown that increasing application of dilute H_2SO_4 has significantly increased the basal area growth (Tamm, et al. 1980). The Norwegian studies consist of field plot experiments where artificial rain has been produced by mixing ground water with H_2SO_4 to pH values from 6 to 2. In one experiment with Scots pine, increased height and diameter growth were observed in 1976 and 1977 at the plots supplied with 250 mm of water per year of pH 3, 2.5, and 2. In 1979, however, the most acidified plots showed significantly less growth than the other experiments (Tveite, 1980).

Although acidification seems to temporarily increase the nitrogen availability in the soil, the increased deposition of inorganic nitrogen from the atmosphere is probably more important for growth increase (Wood and Bormann, 1975; Abrahamsen, 1980). The nitrogen deposition is currently 5 to 10 kg N/ha/year in southern Scandinavia. As nitrogen is the main growth limiting element in forests, increased deposition of nitrogen will most likely increase forest growth. Increased growth combined with increased leaching of magnesium, calcium, and potassium may produce future deficiencies in these elements (Abrahamsen, 1980).

Field investigations on possible growth effects in boreal coniferous forests receiving acid precipitation have been inconclusive. Jonsson and Sundberg (1972) classified areas in southern Sweden as relatively resistant to acid rain and relatively susceptible to acid rain, and compared growth trends in both areas by measuring annual rings from groups of trees which were otherwise nearly identical. They found a statistically significant difference and "found no reason for attributing the reduction in growth to any cause other than acidification." These results, however, have not been confirmed by Norwegian researchers (Abrahamsen, et al. 1976; Strand, 1980.)

A number of possible effects of acid precipitation on the biological and biochemical processes in forest soil have been identified, and are reviewed by Tamm (1976) and Alexander (1980). Among these are:

1. Changes in soil microbiological populations, such as decreases in bacteria and subsequent increase in soil fungi. Effects on humus decomposition have been noted.

2. Nitrogen turnover, which is connected to organic matter decomposition. Effects, mostly reductions, have been observed in N-mineralization, nitrification, and N-fixation.

CONCLUSIONS

- Atmospheric transport of sulfur compounds and other acidifying components has caused extensive regional acidification of water courses in sensitive areas, both in Europe and North America.
- The regions affected by acidification are presently increasing in area. Lakes in these areas are now characterized by low pH, high contents of sulfate, and high concentrations of several metals, notably aluminum, which is leached from the catchments under impact of acid precipitation.
- Acidification of inland waters has had major effects on life in rivers and lakes. Investigations have shown that all types of organisms in the freshwater ecosystem are affected by acidification, ecosystem structures are simplified, and the lakes probably have become poorer in nutrients.
- A prominent feature of regional water acidification is the extensive loss of fish populations, caused primarily by reproductive failure. Physiological stress and fish kills are caused by toxic combinations of water acidity and high aluminum content.
- Acid precipitation and dry deposition of acidifying components interact with vegetation surfaces, leading both to adsorption and leaching from the foliage. Soils which are impacted by acid deposition, lose basic elements during the neutralization process. These include, in particular, calcium and magnesium which are important nutrients for plant growth. Aluminum also is leached from soils under acidification, with toxic consequences for aquatic life. The mobile sulfate ion provided by acid deposition plays an essential role for transport of cations in the soil solution.
- The possible negative effects on boreal forest growth of nutrient deficiency caused by cation leaching seem to be offset at least in the short term by the fertilization effect by nitrogen compounds in acid precipitation. Little is known of the time required for possible long-term effects, for instance, of magnesium deficiency to become extensive. Several important biological and biochemical processes in soils are affected by acid deposition.

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CHANGING pH AND METAL LEVELS IN STREAMS AND LAKES IN THE EASTERN UNITED STATES CAUSED BY ACIDIC PRECIPITATION

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ABSTRACT

The average pH of precipitation falling east of the Mississippi River is less than 5.0, locally less than 4.0. The pH of rain and snow has decreased locally up to 0.75 units in the last 25 years. Aquatic ecosystems in many large areas are vulnerable to this acidic precipitation because of geologic and soil conditions. Time studies of pH and alkalinity for sensitive surface waters exist for North Carolina, Pennsylvania, the Adirondack Mountains area of New York, New Hampshire, and Maine. The duration of observations ranges from 2 to 50 years. All studies indicate generally decreasing pH and alkalinity. Precipitation event and snow melt studies of pH and alkalinity in Virginia, the Adirondack Mountains of New York, and at Hubbard Brook, N.H. indicate that only mildly acidic (5 to 6) or circum-neutral (6 to 7+) streams may undergo severe pH depression (1 to 3 pH units). Heavy metal data for lakes and streams are sporadic and widely distributed. Precipitation heavy metal data are even rarer. Paleolimnologic data from New England and the Adirondack Mountains of New York indicate increasing atmospheric fluxes of many metals (especially Pb and Zn). Increases in Pb are apparently related to atmospheric particulates. Zn is chemically more mobile and in strongly acidified ($pH < 5.0$) aquatic ecosystems there is a net loss from the system. Increases in Al in surface waters in the Adirondack Mountain area correlate strongly with pH decreasing below 6.0. Leaching of Mn, Zn, and Ca from acidified soils and lake sediments suggest that concentrations of these metals have increased in surface waters over the last 50 years and may now be decreasing because of impoverished soils.

INTRODUCTION

Considerable literature evaluates the impact of anthropogenic activities within drainage basins on surface water quality (e.g., Likens, et al. 1970) and on sediment chemistry (e.g., Shapiro, Edmondson, and Allison, 1971; Bradbury and Megard, 1972) in the United States. Most of these studies focused on gross pollution or large disturbances of a steady state. Only recently (Schofield, 1976; Davis, et al. 1978; Norton, Hess, and Davis, 1980) has attention been focused on aquatic ecosystems with no drainage basin disturbances; there it is possible to isolate the effects of acidic precipitation and associated metal loading on surface water quality and sediment chemistry.

Polluted air and thus polluted precipitation are not inventions of 20th century industrialized society (TeBrake, 1975; Smith, 1872). However, only recently has the regional (even hemispheric) scope of atmospheric and precipitation pollution been recognized (in the U.S., Cogbill and Likens, 1974); in Scandinavia, Oden, 1976; in Greenland, Cragin, et al. 1975). One of the few positive effects of thermonuclear bomb testing has been the documentation of global dispersal of reaction products (e.g., Cs^{137} and Sr^{90}) (Toonkel, 1980) and obviously other pollutants.

Historical data on the pH of precipitation and surface waters in the United States prior to significant air pollution are non-existent. Before the mid-1950's, pH measurements were generally made with colorimetry, making comparison with modern (electrode) measurements difficult and somewhat ambiguous (Spikkeland, 1977; Boyd, 1980). Even measuring pH with electrodes is difficult because of the low ionic strength of precipitation and some surface waters (Galloway, et al. 1979). Early measurements of pH of surface waters were performed downstream from waters which would respond to changes in precipitation chemistry and in lakes subject to direct human influence. Precipitation pH measurements, until the establishment of the National Atmospheric Deposition Program network (NADP, 1980), have been non-regional, short-lived, and difficult to compare for numerous reasons. Consequently, strict correlation of the changing pH of precipitation with surface water pH changes is not generally possible.

Similarly, Lazrus, et al. (1970) were the first to produce data for heavy metals in precipitation. More recent studies are short-lived, non-regional, and generally not comparable because of differing collection or analytical techniques (Galloway, et al. 1979, 1980). However, there is no doubt that concentrations have increased on a local (Bertine and Goldberg, 1971) and hemispheric scale (Herron, et al. 1976). Chemical profiles in recent deposits in ombrotrophic peat bogs (Livett, et al. 1979) suggest an increased atmospheric concentration of certain metals, notably Pb and Zn.

Empirical studies along modern environmental gradients have been undertaken because of the lack of historical data for pH of, and metals in, precipitation and surface waters. The basis of these studies is that if one is able to control variables in certain environmental parameters, it is possible to assess time dependent processes related to increased atmospheric deposition of acids and metals on (a) acidification of streams, (b) acidification of lakes, (c) acidification of soils, and (d) mobilization and/or accumulation of metals. Simple unambiguous conclusions can be reached by these types of studies. For example, small oligotrophic non-dystrophic lakes, in the absence of watershed disturbances, become acidified only where the precipitation is acidic.

VULNERABILITY/SENSITIVITY

For significant changes in pH or metal concentrations to occur in surface waters in response to changes in the chemistry of precipitation, the aquatic ecosystem must be vulnerable and sensitive.

Virtually all of the eastern half of the United States is receiving precipitation with an annual average weighted pH less than 5.0. The northeastern States (New England, New York, New Jersey, and Pennsylvania) are receiving precipitation with an annual pH less than 4.5. Precipitation with a pH less than 4.0 is common even as far northeast as central Maine; pH's less than 3.0 have been recorded (NADP, 1980; Likens, 1976). Acidic precipitation also occurs in Washington (Gillion and Horner, 1977) and California (McColl, 1980; Morgan and Liljestrand, 1980) but the geographic distribution on the west coast is relatively restricted at present. Thus, half of the surface waters of the United States are potentially vulnerable to low pH of the precipitation.

The response to additional acidic precipitation is a measure of sensitivity. Chemically, sensitivity measures the proton assimilative capacity and assimilation kinetics of the ecosystem. In the absence of anthropogenic disturbances, sensitivity is controlled by the soils and bedrock geology. Assimilation may occur by:

A. Solution of rocks/minerals such as

$$Al(OH)_{3_{x1}} + 3H^{+}_{aq} = Al^{+++}_{aq} + 3H_2O_{aq} \text{ and}$$

$$CaCO_{3_{x1}} + H^{+}_{aq} = Ca^{++}_{aq} + HCO^{-}_{aq} \text{ or}$$

B. Loss of alkalinity by such reactions as

$$HCO^{-}_{3_{aq}} + H^{+}_{aq} = H_2CO^{0}_{3_{aq}} \text{ and}$$

$$H_2PO^{-}_{4_{aq}} + H^{+}_{aq} = H_3PO^{0}_{4_{aq}} \text{ and}$$

C. Cation exchange reactions such as

$$Al(OH)^{-}_{4_{aq}} + 2H^{+}_{aq} = Al(OH)^{+}_{2_{aq}} + 2H_2O_{aq} \text{ or}$$

$$CA^{++} - Organic_{solid} + 2H^{+} = H^{+}_{2} - Organic_{solid} + Ca^{++}$$

Neutralization of acidic precipitation in soils primarily uses mechanisms A and C. Neutralization of surface waters is dominated by mechanism B.

McFee (1980) and Norton (1980) have developed maps based on soils and geologic criteria, respectively, showing the distribution of sensitive areas in the eastern United States. These maps enable prediction of impact caused by acidic precipitation. A portion of one of these maps is shown in Fig. 1. It demonstrates the scale of variability of sensitivity to be expected in a geologically complex area. Similar results prevail for the soils analysis. Complete coverage depicting geologically sensitive areas for all of the eastern United States is given in Hendrey, et al. (1980).

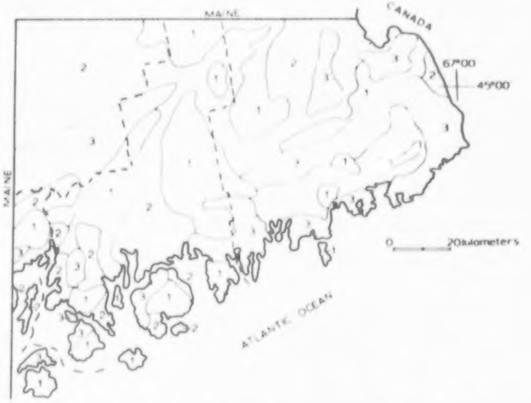

Figure 1. — Sensitivity of part of coastal Maine to acidic precipitation. Sensitivity is indicated as follows: 1 = surface waters will show measurable decline in pH and alkalinity; 2 = surface waters will locally show measurable decline in pH and alkalinity, particularly during precipitation episodes; 3 = surface waters will only undergo pH and alkalinity decline during periods of overland flow. Dashed lines are county boundaries.

pH OF FRESH SURFACE WATERS

Although pH data are abundant for lakes and streams, they are generally not suitable for temporal studies of changing pH for the following reasons:

1. Most of the studies on streams and lakes have focused on populated areas where local anthropogenic activity may dominate the chemistry and where sensitivity has been lost by virtue of upstream heterogeneous soils and geology (Fig. 1). For example, some of the longest series of data are U.S. Geological Survey gauging stations which are not located on first, second, or third order streams.

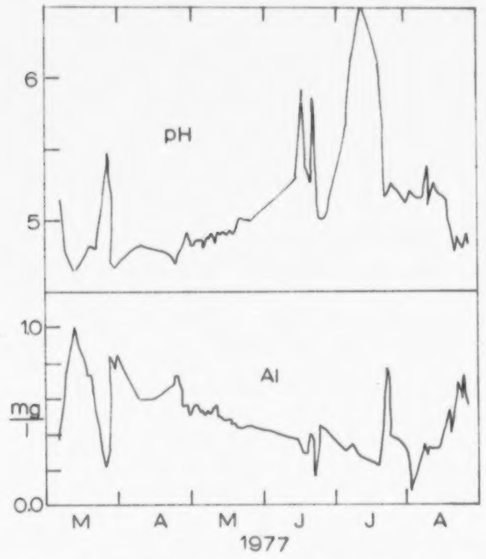

Figure 2. — Short-term pH and Al variations in an Adirondack Mountain stream, New York. Generalized from Schofield (1977). Note reciprocal relationship.

2. Episodic excursions of pH in streams are common. Burns and Galloway (in Hendrey, et al. 1980) in Virginia, Schofield (1977, 1979) in the Adirondack Mountains, N.Y. (Fig. 2), Hornbeck, et al. (1977) at Hubbard Brook, N.H., and Haines and Norton in Maine (unpubl. data) have demonstrated that the pH of unbuffered streams can oscillate as much as 2 pH units over a few days, depending on the relative proportions of overland and groundwater flow involved in stream discharge.

3. Changing land use has been responsible locally for short to long-term changes in stream and lake pH (Likens, et al. 1970; Rosenqvist, et al. 1980).

Consequently, studies based on temporally paired stream pH are suspect. Arnold, et al. (1980) obtained paired data for 314 streams with pH and alkalinity measured twice (more than 1 year apart). Of the 314 streams 107 (34 percent) showed a decrease in pH, alkalinity, or both. Therefore, 66 percent were constant or increased in both pH and alkalinity. Although Arnold, et al. (1980) claim that 34 percent have been acidified by acidic precipitation, it is far more likely they are randomly more acidic on one individual measurement. Only with a large sample of randomly distributed pairs could one hope to detect a statistically meaningful temporal drift in pH. High pH surface waters are particularly susceptible to large random variations because of factors other than changing precipitation pH. Ideally, low alkalinity streams should be sampled at closely spaced intervals over a long period of time such as has been done by workers in Sweden (Oden, 1976) (Fig. 3).

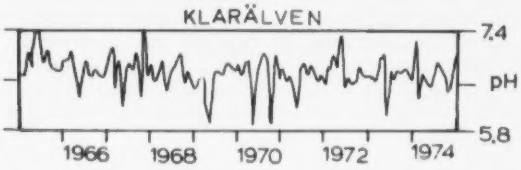

Figure 3. — Long-term pH variation in an southern Sweden stream. Generalized from Oden. (1976).

Because of the volume and long-term flushing characteristics of lakes, they may integrate, smoothing out the pH variations caused by the changing pH of precipitation. Nonetheless, variations in lake water pH may occur which are unrelated to long-term changes in the pH of precipitation. These variations may be caused by:

1. Seasonal changes in surface runoff/groundwater flow.
2. Photosynthesis/respiration in the water column.
3. Sediment/water interaction.

Consequently, comparison of historic data for pH trend analysis for lakes is plagued with the same general problems as for streams, perhaps to a lesser degree. Again, numerous paired data, randomly distributed in time (and separated by as much time as possible) and randomly distributed with respect to all variables, should reveal pH trends, particularly for low alkalinity waters.

Fig. 4 shows a systematic shift in 27 paired North Carolina stream pH's, separated by at least 15 years. Twenty-three of 27 streams were more acidic in 1979 than in 1960-64. Random variations should distribute the points equally about the diagonal line (Fig. 4). Similar relationships exist for 35 streams showing decreased alkalinity.

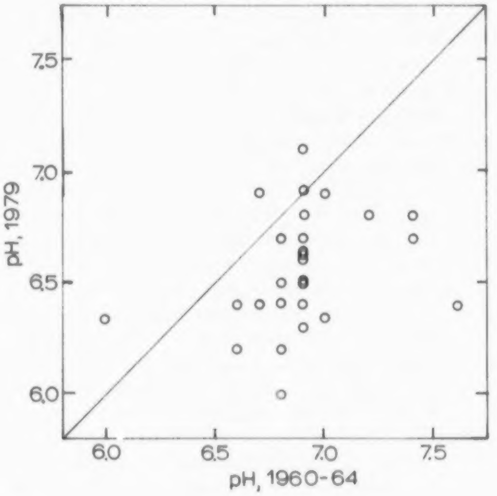

Figure 4. — pH (1979) versus pH (1960-1964) for North Carolina streams. Diagonal line is the locus of no change (from Hendrey, et al. 1980).

In the Adirondack Mountains of New York, Schofield (1976) selected a group of high altitude lakes for comparative studies. Most of these were located in sensitive terrain and analysis revealed a marked decline in pH of lakes. In 1929-37, 5 percent of 217 lakes had a pH below 5.0; in 1976, 51 percent had a pH below 5.0. Hendrey, et al. (1980) analyzed paired data from lakes and streams in New Hampshire and found relationships there to be similar to those in North Carolina. Davis, et al. (1978) studied 1,368 low elevation lakes in Maine and reported a similar trend (Fig. 5). Of 37 low elevation oligotrophic lakes in Maine (Davis, et al. 1979) 31 had decreased 0.2 to 0.7 pH units between 1935-1945 and 1978.

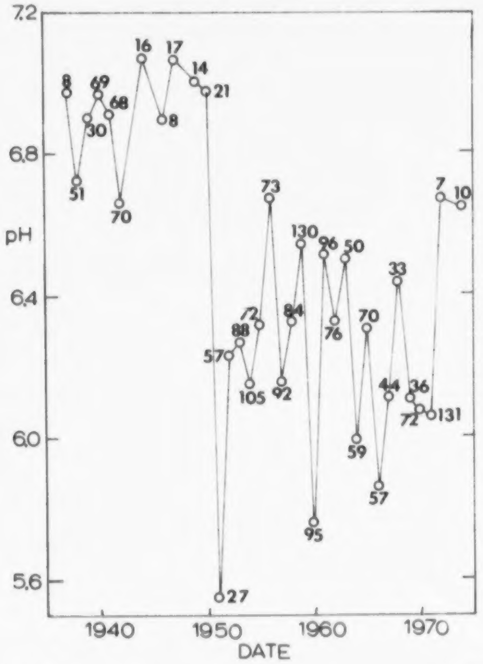

Figure 5. — Annual mean pH's for a data set from 1368 lakes in Maine. Each mean is labeled with the number of pH readings on which it is based (Davis, et al. 1978).

METALS IN FRESH SURFACE WATERS

Recent literature contains abundant data for rivers and lakes on trace metals (other than the major elements Na, K, Ca, Mg, and Si) such as Fe, Mn, Al, Zn, Pb, Cd, etc. However, most of the studies are not useful in determining temporal changes in trace metal content caused by atmospheric deposition. Most of the studies were initiated because of suspected pollution by anthropogenic activities located within the drainage basin. Commonly, the concentrations for these pollutants far exceed the concentrations one would expect to find related to changes (either pH or metal concentration) in the chemistry of precipitation. Additionally, most of these studies are on higher order (third, fourth, or more) streams where effects of changing precipitation pH are less pronounced. Also, the techniques for chemical analysis have evolved rapidly and are not strictly comparable with older results. Only recently has it become possible to analyze directly for some metals in the ug/l range (Zn, Pb, Hg, Cd, Cu, Al) without pretreatment such as extraction or evaporation (Kleinkopf, 1960).

Just as for pH, metal concentrations in surface waters are subject to short-term variations. Consequently, long-term studies of rivers and lakes are necessary to assess long-term changes caused by changing precipitation pH (and consequent changed metal mobility) and atmospheric deposition of metals. To our knowledge, no useful long-term data exist for trace metals for lakes or streams in the United States which enable assessment of precipitation-related changes. Therefore, to anticipate such changes, one must turn to either transect studies or paleolimnologic evidence.

Geographic transects for metals (precipitation-derived or leached) in surface waters of chemically comparable water bodies have not been done in the United States as they have in Norway (Henriksen and Wright, 1978) where pH of precipitation relates to trace metal content of low pH lakes. An alternative approach is to evaluate the relationship between pH and the concentration of some metal in a variety of surface waters in a small area receiving relatively uniform composition precipitation. Fig. 6 shows the relationship between Al and pH for lakes at high altitude in the Adirondack Mountains, N.Y. Similar relationships have been observed in southern Norway. From this we might anticipate that Al should increase in surface waters as pH decreased with time (Norton, 1976). Similar relationships should exist for Zn, Mn, and other metals with pH sensitive solubilities (Norton, Henson, and Campana, 1980) and have been noted by Schofield (1976) in an area receiving relatively uniform precipitation but with widely varying surface water pH. Data for this type of analysis must be carefully evaluated because Al (or any other metal) may vary drastically with pH (Fig. 2) because of varying proportions of overland and groundwater flow to lakes and streams.

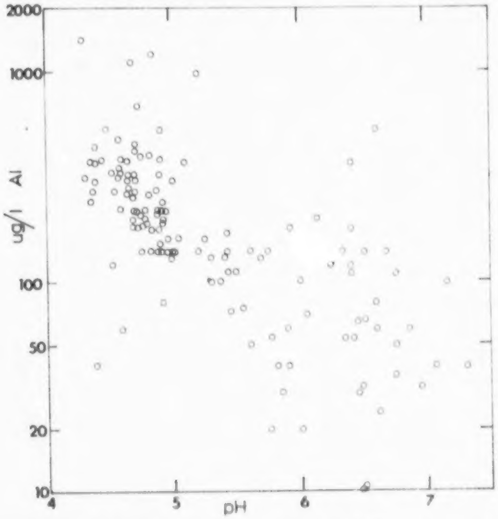

Figure 6. — Aluminum versus pH for 217 high altitude lakes in the Adirondack Mountains, N.Y. Generalized from Schofield (1976).

Paleolimnologic chemical analyses of sediment cores from unpolluted lakes have been used to evaluate changes in metal concentrations in the water or fluxes of the metals through the ecosystem. Although most studies have focused on lakes with drainage-basin sources of metals, several studies have deliberately focused on lakes with undisturbed watersheds except for natural successional changes and natural catastrophes such as fires, pests, floods, etc.

Lake sediments may behave as sinks for certain metals (e.g., Pb). Consequently, an increased flux from the atmosphere will be reflected in concentration profiles in the sediment (Fig. 7). Other metals (e.g., Zn, Lazrus, et al. 1970) although proportionately more abundant in lower pH precipitation (NADP, 1980) may accumulate in non-acidified ecosystems, reach steady state in moderately acidified ecosystems, and decrease in strongly acidified systems (Fig. 7). This corresponds to increasing, constant, and decreasing concentrations, respectively, in current sediments. The ubiquitous and concurrent rise in heavy metals in sediments in relatively pristine lakes (Fig. 8) suggests that atmospheric deposition causes the observed changes.

Chemical profiles from strongly acidified lakes (Williams, 1980) (see e.g., Dream Lake, pH = 4.5, Fig. 9) suggest that detritus reaching the lake has been depleted of Ca, implying a temporary elevation of Ca in surface waters until readily leached Ca is removed. (Mn, Cu, Zn, and Mg also decrease, as expected during acidification.) Malmer (1976) noticed a decrease with time in Ca in southern Sweden surface waters as did Thompson in Nova Scotia rivers (1980, mss.). Other workers (e.g., Watt, Scott and Ray, 1979, also in Nova Scotia) found no change in Ca with a decrease in pH in certain lakes. Schofield (1976) found in 219 lakes a decrease in Ca with decrease in pH. Presumably, these variations represent different stages in the release of Ca from soils during acidification. Hanson (1980) found

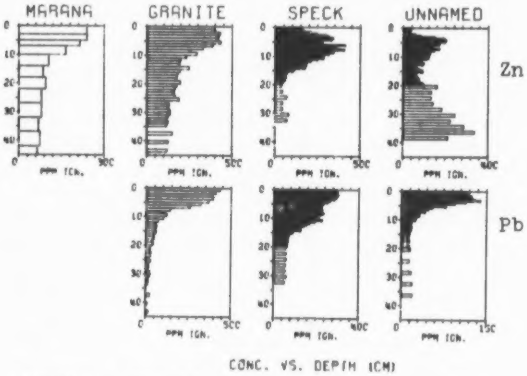

Figure 7. — Pb and Zn profiles for sediment from four New England Lakes: circum-neutral (Maranacook, Maine); slightly acidic (pH. 5-6) (Granite, N.H.); and acidic (pH < 5) kettle pond ("unamed" Pond, Maine). (Norton, et al. 1980b).

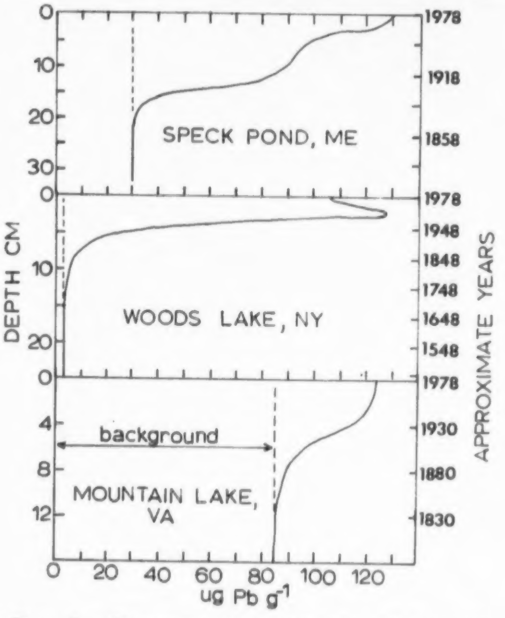

Figure 8. — Pb concentrations in sediment from Mountain Lake, Va. (Galloway, et al. 1980), Woods Lake, N.Y. (Galloway and Likens, 1979), and Speck Pond, Maine (Davis, et al. 1979).

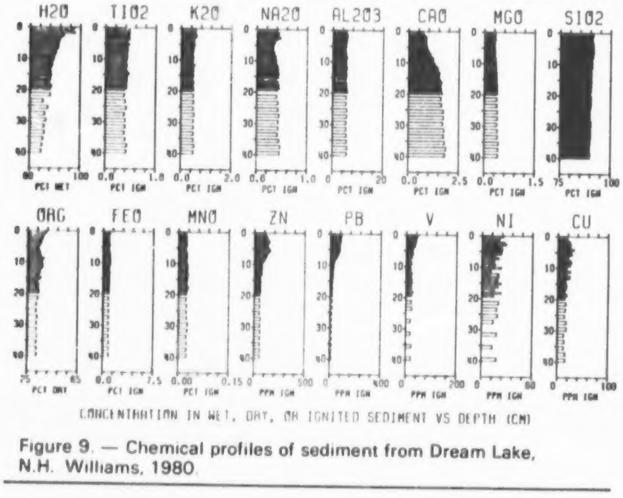

Figure 9. — Chemical profiles of sediment from Dream Lake, N.H. Williams, 1980.

decreasing Ca, Mn, Mg, and K in soil litter subjected to increasingly acidic precipitation on a transect from southern Vermont (site 1) to the Gaspe' Peninsula, Quebec (site 14) (Table 1). Steady-state release (pre-air-pollution) would be followed by increased Ca in surface waters with lowered pH, followed by a decrease in Ca to a new equilibrium steady state release, the level depending on the new pH.

Table 1. — Chemistry of forest litter from high altitude fir forests. Sample sites range from southern Vermont (site 1) to the Gaspe' Peninsula, Quebec (site 14). The pH of precipitation ranges from about 4.0 to 4.6 Details of collection and analysis are in Hanson (1980).

Site	Dry wt % Ca	Ca/Al	ppm Mn	Mn/Al	Dry wt% Mg	Mg/Al
1	0.370	1.03	110	0.31	0.050	1.39
2	0.216	0.27	49	0.06	0.042	0.53
3	0.373	0.63	122	0.21	0.041	0.69
4	0.499	1.00	182	0.36	0.065	1.30
5	0.400	0.68	278	0.47	0.050	0.85
*6	0.653	1.52	373	0.87	0.070	1.63
7	0.301	0.37	270	0.33	0.068	0.84
8	0.494	1.27	364	0.93	0.058	1.49
9	0.654	1.60	259	0.63	0.059	1.44
10	0.628	1.40	297	0.66	0.078	1.73
11	0.814	2.81	424	1.46	0.064	2.21
*12	0.749	0.95	255	0.32	0.050	0.63
13	1.006	2.05	752	1.54	0.070	1.43
14	0.962	2.53	552	1.45	0.063	1.66

*anomalous sites, probably contaminated with mineral soil.

SUMMARY

Acidic precipitation and associated metal deposition in the eastern United States have caused the following changes:

1. Decreasing pH in lakes and streams rendered sensitive by soils and bedrock chemistry.
2. Decreasing alkalinity in the same lakes and streams.
3. Increasing dissolved Al and Ca and probably other cations in acidifying aquatic ecosystems.
4. Increased flux to the aquatic ecosystems of some trace metals (e.g., Pb) and accumulation/steady state/release/net loss from the system of other metals, depending on the pH.

Changes that can be reasonably anticipated with increasing acidification include.

1. Increasing levels of dissolved Al, Fe, and Mn and other major elements (solution of soil minerals).
2. Increasing Ca, Mg, and K (desorption).
3. Increasing levels of metals (Cd, Cu, Zn) whose mobility is increased by lower pH.

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VARIATIONS IN THE DEGREE OF ACIDIFICATION OF RIVER WATERS OBSERVED IN ATLANTIC CANADA

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ABSTRACT

Freshwater bodies in large portions of eastern Canada are adversely affected by acidic precipitation, with resultant damage to fish and other components of the aquatic ecosystem. Concern exists regarding the future degree of acidification in these regions, in the face of increasing emissions of sulfurous and nitrous compounds. This report deals with several questions which arise from such concern: (1) What changes in river water chemistry can be attributed to acid loading? (2) What year-to-year variations in acid loading/response have been observed in river systems? and (3) What fraction of acidification is associated with sulfate deposition?

INTRODUCTION

Freshwater bodies in large portions of eastern Canada are adversely affected by acidic precipitation, with resultant damage to fish and other components of aquatic ecosystems and elevated concentrations of heavy metals. This situation is caused by a combination of circumstances, namely, relatively high rates of acid loading from the atmosphere, and relatively low buffering capacity of the receiving watersheds (Figure 1, from Thompson, et al. 1980). Concern exists regarding the future degree of acidification in these regions, in the face of increasing emissions of sulfurous and nitrous compounds. This report deals with several questions which arise from such concern:

1. What changes in river water chemistry have been observed in eastern Canada that can be attributed to acid loading?,
2. What year-to-year variations in acid loading/response have been observed to occur in river systems?, and
3. What fraction of acidification is associated with sulfate deposition?

"NORMAL" CHEMICAL WEATHERING

Chemical weathering in watersheds that receive normal precipitation is predominantly due to the action of carbonic acid. Carbonic acid is formed by solution of atmospheric carbon dioxide in rain water or surface water:

$$CO_2 + H_2O \longrightarrow H_2CO_3$$

Reactions of carbonic acid with carbonates and silicates can be represented as

$$\begin{aligned} H_2CO_3 + CaCO_3 &\rightarrow Ca^{++} + 2HCO_3^- \\ 2H_2CO_3 + \text{Ca-silicate} &\rightarrow Ca^{++} + 2HCO_3^- + H_2\text{-silicate} \end{aligned} \qquad \text{eq. 1}$$

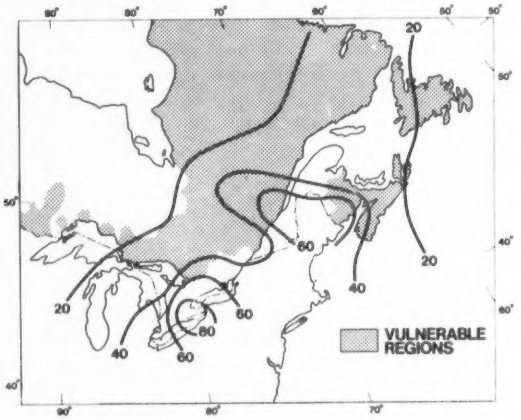

Figure 1. — Atmospheric deposition of hydrogen ion in 1977 (mg / m² a) and soft water regions of eastern Canada.

These show that carbonic acid supplies protons which are exchanged for cations (Ca^{++}, Mg^{++}, Na^{+}, K^{+}) in the crystal lattice of the minerals comprising the soils or rocks, causing the release of cations and bicarbonate ions into solution. The rate at which cations are removed from the watershed (cation denudation rate or CDR) is a measure of the reactivity of the basin or of the rate at which chemical weathering proceeds in the watershed. The rate of production of bicarbonate is correlated with the CDR; indeed, in basins where no other anion (sulfate, for example) is a weathering product, the two rates are equal. Where the rocks are resistant, the bicarbonate concentration will be low, as will concentrations of all other ions derived from weathering. Many of the watersheds of eastern Canada are in this class, for they are composed of resistant granitic and siliceous bedrock from which extensive glaciation has stripped away any younger, calcareous deposits that may have existed.

Another consequence of the carbonic acid system is a positive correlation between pH and alkalinity (bicarbonate concentration) in runoff water. This is a manifestation of the fact that for water in equilibrium with the atmosphere, the product of the concentrations (activities) of the hydrogen and bicarbonate ions tends to be a constant:

$$[H^+]\,[HCO_3^-] = \text{constant} \qquad \text{eq.2}$$

It follows that pH, alkalinity, and CDR are positively correlated; high values of each are characteristic of hard water, with low values for soft water.

Chemical weathering by carbonic acid always increases the alkalinity of runoff water above that of the precipitation which falls on the basin (notwithstanding the effects due to concentration by evaporation in the watershed). Consistent with equation (2), normal or "pure" rain has a pH of about 5.6. Accordingly, the water discharge from a basin that receives such rain must have a pH higher than 5.6.

WEATHERING BY ACID PRECIPITATION

The fact that normal rain has a pH of about 5.6 is a consequence of the carbonic acid system. The pH of acid rain is lower than 5.6 because of the presence of strong acids such as sulfuric acid. In a watershed the rain-borne strong acids serve as a ready source of protons for exchange with cations in weathering processes:

$$\begin{aligned} &H_2SO_4 + CaCO_3 \rightarrow Ca^{++} + SO_4^{=} + CO_2 + H_2O \\ &H_2SO_4 + \text{Ca-silicate} \rightarrow Ca^{++} + SO_4^{=} + H_2\text{-silicate} \end{aligned} \qquad \text{eq. 3}$$

These proton-cation exchanges are the same as when carbonic acid is involved (equation (1)), but bicarbonate alkalinity is not a byproduct. Accordingly, weathering by acidified precipitation tends to produce discharge water of pH relatively low compared to that resulting from the action of normal rain.

In general, the weathering action of acid rain should be considered to be of both the carbonic acid and strong acid types, especially since in some basins the free protons of the strong acid will be supplied at a rate insufficient to match the CDR. Therefore, in any watershed, the degree to which pH is depressed in the discharge water depends on the strength of strong acid in the precipitation falling on the basin, and on the resistance of the rocks. The aquatic regimes in basins composed of easily-weathered minerals may show little adverse effects from acid precipitation because of continued dominance of bicarbonate alkalinity. However, hard-rock watersheds with low potential buffering capacities will show relatively large pH changes corresponding to a given change in rain acidity.

The rate of arrival of strong acids in precipitation must be considered with respect to the rate of chemical weathering; thus, even a relatively reactive watershed might be temporarily adversely impacted because the rate of acid input momentarily overwhelms the rate of weathering, e.g., melting of snowpacks.

WATER CHEMISTRY CHANGES

It follows that time histories of pH in soft water drainage basins with little local anthropogenic influence can be interpreted in terms of trends in the acidic strength of the rain. Moreover, a change in pH of the discharge water should be accompanied by a corresponding difference in the concentration of sulfate ion, when the rain acidity is due primarily to sulfuric acid.

The question of specific contribution of sulfuric acid to acidification can be addressed by noting that the sum of the concentration of bicarbonate ion and twice that of the sulfate ion tends to be constant at any sampling location in an aquatic system:

$$[HCO_3^-] + [SO_4^{=}] = \text{constant (CDR)} \qquad \text{eq. 4}$$

This means that if sulfate increases in water because of a rise in the sulfurous acidity of rain, then alkalinity must decrease; and if the sulfate concentration falls, alkalinity rises. This relationship is consistent with earlier remarks regarding weathering by acidic precipitation, is derived from equations (1) and (3), and is true as long as the CDR is constant. In addition, equation (4) is not valid if the sulfate concentration exceeds an upper limiting value that is related to the CDR. However, in general, the relationship is a powerful tool for analyzing or predicting changes in water chemistry that can be attributed to changes in sulfate loading. The strong/weak acid relationship (4) can be combined with the acidity/alkalinity relationship (2) to yield

$$[H^+] = \frac{[H]_0\,[HCO_3^-]_0}{[HCO_3^-] + 2[SO_4^{=}]_0 - 2[SO_4^{=}]} \qquad \text{eq. 5}$$

Here the acidity is related to prior or initial values of the acidity and alkalinity (subscripts), and to the change in sulfate concentration.

Equation (5) is useful for calculating expected values of the acidity corresponding to observed or potential changes in sulfate concentration, as long as the sulfate concentration does not exceed the limiting value given by

$$[SO_4^{=}]_L = 1/2\,[HCO_3^-]_0 + [SO_4^{=}]_0 \qquad \text{eq. 6}$$

If the rain-borne sulfate loading is sufficiently high, then the sulfate concentration in the runoff water may exceed $[SO_4^{=}]_L$, which means that the rate of sulfate discharge exceeds the CDT of the basin. In this case the chemical weathering processes can be considered to be entirely of the strong acid type, and the acidity of the runoff water to depend primarily on the difference between the sulfate loading rate and the CDR. It is equally correct to consider that the acid precipitation neutralizes the alkalinity that would have been produced by only carbonic acid weathering, and that the pH of the runoff water is a function of the excess acidity. In any case, equations (4) and (5) would not be valid for such instances of strong acidity. A further consequence of equations (2), (4), and (6) is that a "pristine" pH can be calculated for those watersheds where the present sulfate content is caused entirely by atmospheric deposition, that is, where there is essentially no sulfate deriving from the minerals. The original bicarbonate concentration would be

$$[HCO_3^-]_{0.0} = 2\,[SO_4^=]_L = [HCO_3^-]_0 + 2\,[SO_4^=]_0$$

while the original pH would be

$$pH_0 = pK + \log[HCO_3^-]_{0.0} \qquad \text{eq. 7}$$

where K is the constant in equation (2). Again, such calculations may be made with the provisos that the CDR is constant over the years, and that the precipitation acidity is essentially due only to sulfuric acid.

It is not yet possible to quantify directly the influence of the atmospheric deposition of nitrogen compounds on acidification of water systems because ammonia and nitrate are value nutrients in terrestrial and aquatic regimes, and their chemistry is uncertain. As it happens, however, most of the acidification observed in eastern Canadian aquatic regimes can be attributed to sulfate deposition.

EXAMPLES OF ACIDITY CHANGES

River water chemistry data are compared here to demonstrate the influence of acidic precipitation in a few watersheds in Nova Scotia and Newfoundland. In each case, the parameters used for comparison are discharge-weighted annual mean values derived from individual samplings, usually at monthly intervals, within each year. Most of the information was collected by Canada's National Water Quality Monitoring Program that began in 1961; a valuable contribution was also made in 1954/1956 in the Atlantic Region by a monitoring program of Water Survey of Canada.

Generally decreasing values of pH have been observed in the rivers and lakes in the Atlantic Provinces and reported by Thompson (1980), Thompson, et al. (1980), and Watt, et al. (1978). Typical histories exist for the Tusket and Medway Rivers that are located in southern Nova Scotia (Figure 2); there the mean pH decreased by 0.7 units in the time interval 1954/1955 through 1973 (Figure 3). Because the CDR of each basin is essentially unchanged, the observed increase in acidity of the discharge water must be due to an increase in the rate of acid loading. Accordingly, the pH decreases should be accompanied by increases in the sulfate concentration in the runoff water from each basin. Listed in Table 1 is information necessary to assess pH-sulfate interdependency for the two Nova Scotia rivers. Of immediate interest are the increases in concentration of non-marine (or sea salt corrected) sulfate between 1954/1955 and 1972, and the close agreement between the pH values observed in 1972 and those calculated from equation (5). These figures show that the observed acidification is primarily caused by an increase in sulfate deposition. Closer inspection of the ratios of the calculated to observed changes in hydrogen ion concentration shows that sulfurous acidity accounts for at least two-thirds of the observed effect. It follows that if sulfate deposition to a watershed is increased, then the acidity of the runoff water is increased.

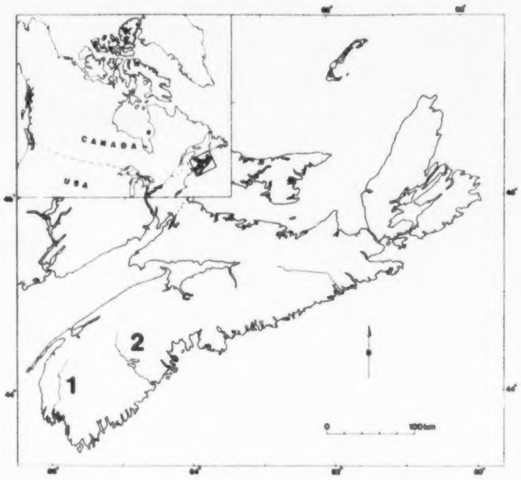

Figure 2. — Location of the Tusket River (1) and Medway River (2) in Nova Scotia.

Table 1. — Some significant values of acidity and sulfate concentration in two Nova Scotia rivers.

		Tusket R.	Medway R.
1954/5	pH	5.14	5.67
	HCO_3^- mg/l	1.84	2.65
	$SO_4^=$ mg/l	2.32	1.71
1972	$SO_4^=$ mg/l	3.36	2.88
	pH calculated*	4.60	5.31
	pH observed	4.48	5.12
	$\frac{\Delta H^+ \text{ calculated*}}{\Delta H^+ \text{ observed}}$	0.76	0.65
	$[SO_4^=]_L$ mg/l	3.77	3.79
	pH_0	6.01	6.12

* calculated from equation (5)

Estimates of the limiting sulfate concentration beyond which pH is a function primarily of excess acid loading were calculated according to equation (6) and included in Table 1. The value of 3.8 mg/l sulfate was obtained for the two basins, implying essentially identical geology. This limiting value was closely approached in 1972 in the Tusket River, and was equaled there in 1973. In both years, sulfate concentration was about 0.5 mg/l higher there than in the Medway River, suggesting that the rate of acidic loading was lower in the Medway watershed than in the Tusket. Such a difference in loading could be expected because of the locations of the watersheds with respect to the continental source regions (Figure 2).

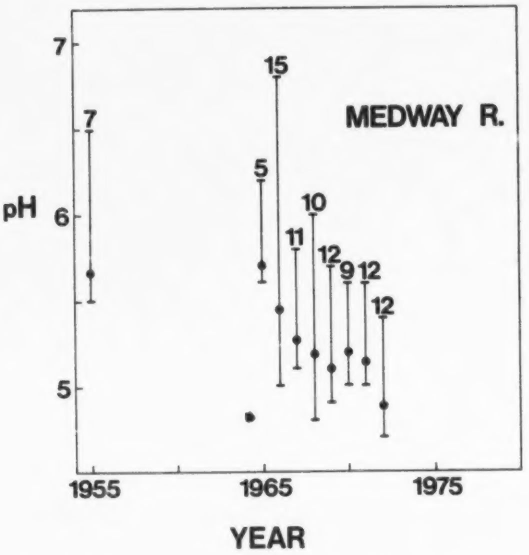

Figure 3. — Summary of pH observations in (a) Tusket River and (b) Medway River. The range and mean values, and the number of observations are given for each year.

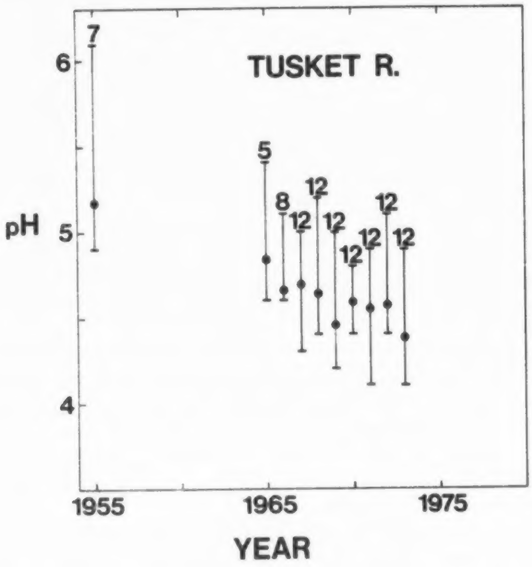

Figure 4. — Annual values of pH and sulfate in the Rocky River, Newfoundland.

Included in Table 1 are the calculated values of the pristine pH for the two basins, estimated from equation (7). How accurate these figures are is not known, because of the unknown influence of nitrogen acidification. However, pH values of 6.0 and 6.1 serve to underscore the fact that, in general, watersheds that today are markedly impacted by acid loading are those which had very little buffering capacity initially, and therefore had a relatively low initial pH.

These temporal and spatial variations in river chemistry are due to variations in acid *deposition* on the watersheds, independent of the rate of emissions at the sources. It is to be expected that natural variability in climatic factors could cause significant variability in atmospheric deposition rates in any basin, especially in fringe areas. This point is illustrated by the history of mean pH and sulfate in the Rocky River, located in southeastern-most Newfoundland (Figure 4). The observations show not only the general upward trend of both acidity and sulfate concentration in the river water, but also simultaneous occurrences of relative maxima and minima of the two properties. Of particular interest are first, the 1973 data, which show record high sulfate and record low pH values, and second, in subsequent years, a return to higher pH levels in association with a decline in sulfate concentration. Some of the year-to-year changes are too large to be due to possible variations in emissions at the sources, and must therefore indicate significant differences in deposition caused by variations in climatic factors.

The fact that pH in the Rocky River rises when the sulfate loading decreases is an important point, for it illustrates that the geochemical response to acid loading in a basin is a dynamic one. Thus relationships such as that given by equation (5) can be used to measure or predict not only the course of acidification due to increasing sulfate loading, but also that of natural recovery which would be associated with reduced sulfur dioxide emissions.

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RESPONSES OF FRESHWATER PLANTS AND INVERTEBRATES TO ACIDIFICATION

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ABSTRACT

The biota of acidic, oligotrophic, clear waters often are similar. The phytoplankton Dinophyceae, and to a lesser extent Chrysophyceae tend to dominate. Production of 25 Shield lakes (pH 6.1 to 7.1) ranged from 25 to 240 mg $C m^{-2} d^{-1}$. Published values for acidic lake production are bracketed by this range. Both biomass and production appear to be controlled by the availability of phosphorus rather than pH *per se*. We found little evidence of possible C limitation in lakes susceptible to acidification $[H^+]$ and biomass density in lakes do not appear to be directly related, as illustrated by the whole-lake manipulations of $[H^+]$ and total phosphorus (TP). These studies, however, do not examine effects of acidification on the whole lake-watershed system. It is suggested that watershed acidification processes such as leaching of Al may reduce TP loading to lakes, even below pH 5. Zooplankton community biomass appears to be reduced at low pH and small-bodied forms may dominate. Among the zoobenthos, biomass does appear to be reduced in some lakes but not others. Various studies found shredders, collectors, and scrapers to be reduced more than raptorial species. We hypothesize that removal of fish predation on benthos allows a relative increase in the invertebrate predators, reduction of herbivores (chironomids are relatively abundant), and the subsequent increase in benthic algae observed in many waters.

INTRODUCTION

Over the past decade, numerous studies of the biota of waters acidified by the deposition of strong acids have described certain changes in communities of fish, zooplankton, bottom fauna, phytoplankton, benthic algae, rooted aquatic plants and mosses, and benthic decomposers. We will point out where diverse responses to acidification have been observed and suggest ecological ramifications of certain of the more common observations which may be useful in interpreting observed changes. Other recent reviews on the effects of acidification on biota are provided by Almer, et al. (1974, 1978), Dochinger and Seliga (1976), Hendrey, et al. (1976), Leivestad, et al. (1976), and Harvey (1980).

PHYTOPLANKTON COMMUNITY COMPOSITION

Regional surveys of Scandinavian lakes of pH ~4.0 to 7.0 have demonstrated that the numbers of species in phytoplankton communities is reduced in acid lakes, especially over a pH range of 6 to 5 (Almer, et al. 1978). Species from all classes are lost although data derived from regional surveys suggest that proportionally more species of Chlorophyta disappear (Almer, et al. 1974). More intensive collections from Adirondack lakes confirm this observation (Figure 1). Biomass of phytoplankton communities of non-acidic oligotrophic lakes are typically dominated by Chrysophyceae (Schindler and Holmgren, 1971) or Bacillariophyceae (Duthie and Ostrofsky, 1974). This pattern changes as lakes acidify (Figure 2). Community structures of four acidic lakes near Sudbury, Ontario were compared to those of 10 non-acidic lakes. Dinophyceae, notably *Peridinium inconspicuum*, replaced chrysophytes and diatoms as community dominants (Yan, 1979). Late summer plankton of 60 Swedish lakes in the pH range 4.60 to 5.45 were dominated by the Dinophyceae, especially *Peridinium inconspicuum* and *Gymnodinium* cf. *uberrium*, while Chrysophyceae dominated in spring. Some acidic lakes, however, were dominated by *Oocystis* (Chlorophyceae) (Almer, et al. 1978).

In some acidic lakes Chrysophyceae remain as community dominants. Three Adirondack Mountain lakes (Woods, pH 4.7 to 5.2; Sagamore, pH 5.0 to 6.4; Panther, pH 5.3 to 2.8) (Hendrey, 1980) show the pattern of reductions in species richness in acidified lakes that is typically observed with losses of species of Chrysophyceae (Figure 1). Chrysophyceae domi-

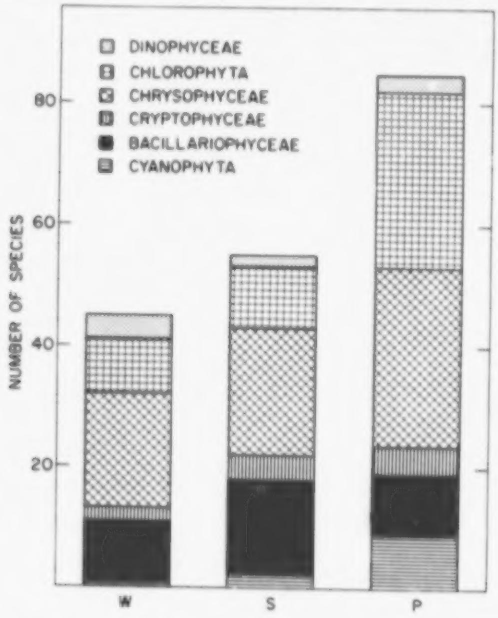

Figure 1. — Total number of phytoplankton species observed in each of three Adirondack Mountain Lakes arranged by classes. (Samples collected biweekly during the ice-free season, monthly during winter from three to five in each lake).

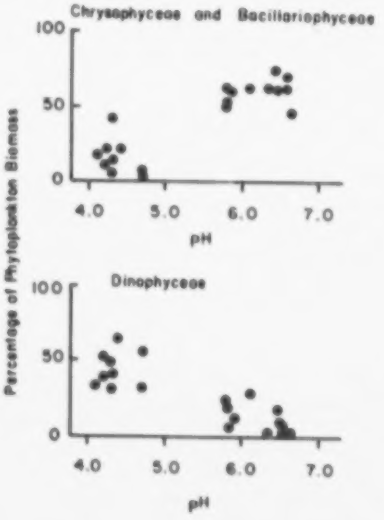

Figure 2. — Distribution of phytoplankton biomass among Dinophyceae, Chrysophyceae and Bacillariophyceae in Sudbury area lakes (before manipulation) and Haliburton area lakes. (These are all softwater oligotrophic lakes. Each point is monthly-weighted, ice-free period mean of biweekly collected, morphometrically weighed euphotic zone composits.)

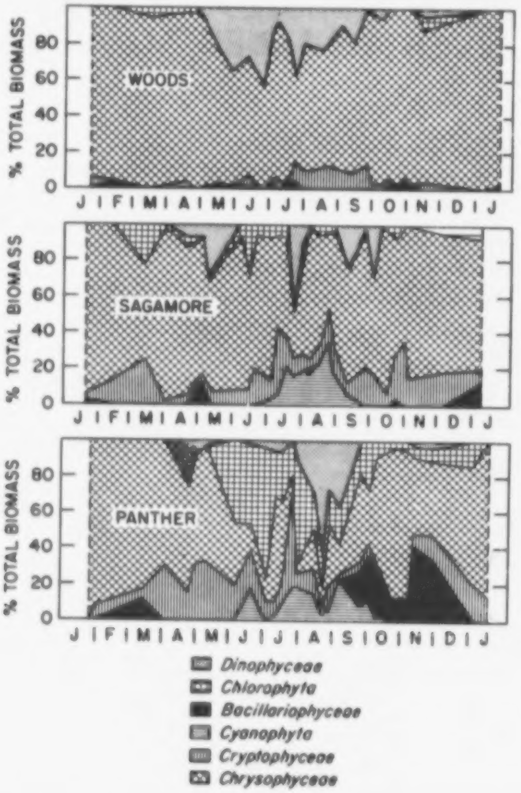

Figure 3. — Seasonal variation in the proportion of total phytoplankton biomass contributed by each taxonomic group in three Adirondack Mountain Lakes (biomass derived by microscopic measurements of cell volume).

nate the biomass of acidic Woods Lake, although Dinophyceae (particularly *P. inconspicuum*) comprise a significant fraction of the biomass in the ice-free season (Figure 3). Perhaps less commonly, Cyanophyta may also be important in acidic lakes (Watanabe, et al. 1973 Chlorophyta outweighing losses of); Conroy, et al. 1976; Kwiatkowski and Roff, 1976). Thus, while it has been observed that Dinophyceae are commonly the dominant phytoplankters in acidic lakes (Almer, et al. 1978; Yan, 1979; Hornstrom, 1979) many exceptions to this pattern exist.

PHYTOPLANKTON PRODUCTIVITY AND BIOMASS

Although many studies show freshwater acidification to be associated with major changes in community structure, only a few have examined the effects on phytoplankton productivity. In 26 Canadian Shield lakes in the pH range 6.1 to 7.1, maximum volumetric productivity ranged from 1.7 to 6.9 mg C m^{-3} hr^{-1} and areal production from 25 to 240 mg C m^{-2} d^{-1}. These data (Harvey, 1980) bracket the published productivity values for acidified lakes, although such data are quite limited (Dillon, et al. 1979; Hornstrom, 1979; Hendrey, 1980).

Carbon Limitation

At equilibrium with atmospheric CO_2 ($P_{CO_2}=10^{-3.5}$ atm.) inorganic carbon concentrations in lakes of pH <5.0 will be under 15 *μm*. Phytoplankton production therefore may be reduced, i.e., limited by available carbon unless carbon concentrations are maintained near saturation by atmospheric invasion of CO_2 or by the respiratory activities of the biota. If, on the other hand, these processes provide dissolved inorganic carbon to the phytoplankton of acidic lakes sufficiently rapidly, then the greater clarity of the lakes may facilitate aerial production rates that are as high or higher than in non-acidic lakes of comparable nutrient status. No evidence has been presented, so far, that lack of carbon might limit phytoplankton production in oligotrophic acidic lakes. In Woods Lake, with DIC values in the range 0.3 to 0.6 mg C l^{-1} (pH 4.7 - 5.1), for example, the maximum volumetric hourly productivity rates (measured biweekly at five depths through the ice free season in 1979) never exceeded 2 percent of the available carbon (DIC measured by gas chromatography, Stainton, 1973), and the maximum daily production per square meter never exceeded 6.4 percent of the DIC available at 10 a.m. as shown in Figure 4 (Hendrey, unpubl. data).

The productivity of Woods Lake is at the upper end of the range of productivity observed in many oligotrophic Canadian Shield lakes (Figure 5). Experimental acidification of ELA Lake 223 reduced DIC from around 1.2 mg.C l^{-1} to the range 0.06 to 0.6 mg C l^{-1} yet phytoplankton photosynthesis was not carbon limited. Short-term bioassays with carbon enrichment did not increase phytoplankton productivity (Schindler, et al. 1980).

With an average daily production rate (which assumes no carbon limitation) of 13.8 *m* moles C m^{-2} (166 mg C m^{-2}), and a mean depth of 8.4 m, phytoplankton production in Clearwater Lake (Dillon, et al. 1979) would consume 1.64 *μ*moles DIC l^{-1} day^{-1}. This represents about 14 percent of the DIC available if carbon concentrations in the lake are at saturation at a lake pH of 4.2. Schindler and Fee (1973) found that in northwestern Ontario lakes in which production was not carbon limited (fertilized lakes), daily production could reduce DIC by 1 to 7 percent. In lakes that were carbon limited, over 25 percent of the carbon available at dawn was consumed during the day. Based on this analysis, Clearwater Lake occupies an intermediate position suggesting that on cloudless days phytoplankton production may be carbon limited.

Phosphorus Limitation

Several studies show that the biomass of phytoplankton is correlated with the supply of phosphorus in both acidic and non-acidic lakes (Schindler, 1971; Schindler and Fee, 1974; Nicholls and Dillon, 1978; Yan, 1979; Hornstrom, 1979). Studies in which the chemistry of whole lakes has been manipulated especially demonstrate the control exerted by phosphorus on phytoplankton biomass. In the fall of 1973 adding base raised the pH of acidic Middle Lake (pH 4.4) to pH ca. 7.0. Total P (TP) levels did not increase; in consequence there was no increase in phytoplankton biomass (Table 1), although species composition shifted from Dinophyceae to Chrysophyceae. TP was experimentally increased in Middle Lake from 1975 to 1977 and a large increase in biomass occurred. TP was also increased in acidic Mountaintop Lake without elevating lake pH and phytoplankton biomass increased. Hydrogen ion concentration also increased by 20 percent after 1 year and by 75 percent after 2 years because of bicarbonate generation from SO_4 reduction in the hypolimnion (Table 1). Comparing biomass in Mountaintop Lake to non-acidified Labelle Lake to which phosphorus was also added, shows the strong dependence of biomass in Labelle on TP and not on pH (Figure 6). Schindler, et al. (1980) slowly acidified Lake 223 in the Experimental Lakes Area (ELA) from pH 6.7 to 7.0 in 1976 to pH 5.7 to 5.9 in 1978 without any apparent change in either productivity or biomass of phytoplankton, or in TP or dissolved P.

These studies indicate that pH change alone does not alter phytoplankton production (to pH>5.7) or biomass (pH>4.4) and that biomass is regulated by the supply of P. However, acid precipitation does alter not only the lake water pH but also watershed processes, particularly the weathering of aluminum from rock and soil, and consequently, possibly the watershed chemistry of nutrients such as phosphorus. Whole-lake manipulations which treat only the lake water have proved very useful in elucidating portions of the lake acidification story. But an ecosystem approach must still be used to interpret the complex phenomena associated with lake-watershed acidification and consequent biological effects.

Table 1. — Total phosphorus (TP), pH and biomass of phytoplankton observed in two Sudbury Experimental Study lakes. Data are ice-free period, monthly weighted means from Dillon, et al. (1979) and Yan (unpubl. data).

Lake		pH	TP ($\mu g l^{-1}$)	phyto-biomass ($mg l^{-1}$)	chlor *a* ($\mu g l^{-1}$)
Middle	1973	4.4	7.3	0.46	0.91
	1974	7.0	7.1	0.16	0.92
	1975-77	6.5	11.6	0.68	2.70
Mountaintop	1976	4.4	43.0	0.72	5.7
	1977	4.5	58.0	2.05	20.1
	1978	5.0	75.0	6.35	64.8

Aluminum and Phosphorus

In studies of 58 oligotrophic lakes in the Swedish west coast region the lowest phytoplankton biomass was found in eight lakes in the pH range 5.1 to 5.6 while biomass was higher on either side of this pH range. This evidence has been used to support the view that nutrient availability is lowest in this intermediate pH range because of Al complexing of P (Almer, et al. 1978).

The solubility of apatite minerals increases in acidic environments, but as soil studies have demonstrated (Brady, 1974), this need not result in increased bioavailability of phosphorus. Watershed acidification elevates concentrations of aluminum in runoff waters

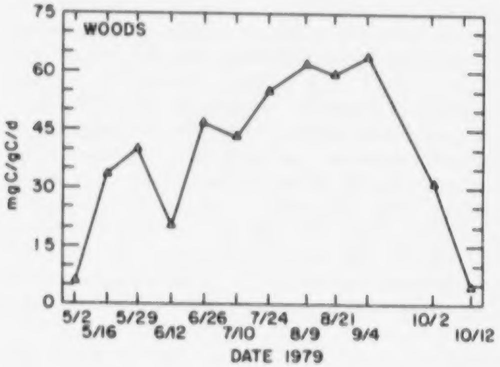

Figure 4. — Ratio of phytoplankton production (mg I m^{-2} d^{-1}) to available carbon (g DIC m^{-2}) in the euphotic zone of Woods Lake (pH 4.7 tp 5.1). Production determed by ^{14}C tracer uptake, at five depths *in situ*.)

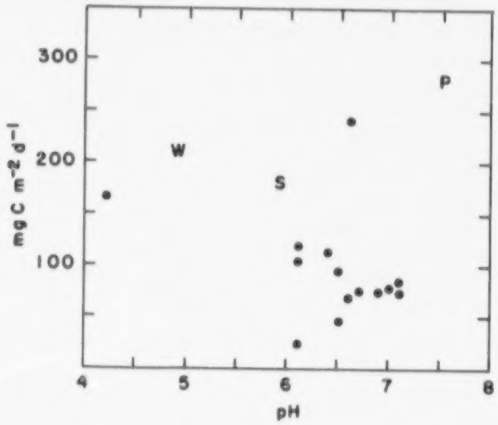

Figure 5. — Maximum observed daily production (mg c m d^{-1}) of phytoplankton in 14 lakes of the Canadian Shield (●) and three Adirondack Mountain lakes (W = Woods, S = Sagamore, P = Panther). Observations were made at least eight times in each lake.

(Cronan and Schofield, 1979). In 37 clearwater Swedish lakes Al concentrations of 0.01 to 0.6 mg l^{-1} occurred with lake water pH 5.5 and were usually less than 0.1 mg l^{-1} at pH 5.5 and above. Dickson (1978) has shown that the removal of phosphorus from lake waters containing additions of 0.5 or 1.0 mg Al l^{-1} is highly dependent on pH, with the maximum removal of P occurring at pH 5 to 6. This does not necessarily mean that waters at pH<5 will contain more soluble P than less acidic water.

The chemistry of Al and its interaction with phosphorus in natural waters is rather complicated and a full discussion is beyond the scope of this review, but several relevant points are worth noting.

1. Driscoll (1980) has found solubility of free Al to be controlled by $Al(OH)_3$, but Al-organic complexes were the dominant form of monomeric Al in surface waters.

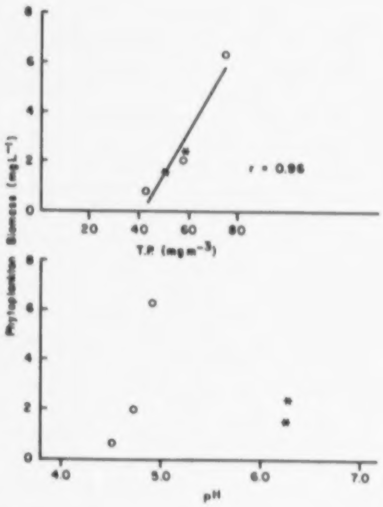

Figure 6. — Phytoplankton biomass in Mountaintop (o) and Labelle(*) lakes versus total phosphorus concentration (TP) and pH. (Average for the ice-free seasons of 1976-1978 in Mountaintop and 1977-78 in Labelle).

2. Dickson (1978) has shown that Al bound to humic substances is unavailable to complex with inorganic P.

3. The minimum solubility of $AlPO_4$ is about 10 μg l^{-1} at pH 6 (see Figure 10-1 in Stumm and Morgan, 1970).

4. Total P concentrations in acidified watersheds have more Al to interact in the softwater, oligotrophic lakes of Scandinavia, the Canadian Shield, and the Adirondack Mountains are typically less than 10 μg l^{-1} (Conroy, et al. 1974) and the fraction of the TP which may be available to phytoplankton is likely to be much lower.

For these reasons it is unlikely that the formation of $AlPO_4$ is an important mechanism in regulating P availability in acidified lakes. On the other hand, it seems likely that aluminum hydroxides (or ferric hydroxide) which have much lower minimum solubilities could remove inorganic P by flocculation, but this mechanism, with respect to acid lakes, has received little attention (Driscoll, pers. comm.).

While Dickson's data (obtained at rather high P concentrations of 50 and 100 μg l^{-1}) indicate less P is removed at pH<5 than at pH 5 to 6 for a lake water with 0.5 mg Al l^{-1}, it is also shown that Al concentrations increase with watershed and lake acidification. Lakes with lower pH in acidified watersheds have more Al to interact in the removal of P, not only in the lake itself but also in the watershed soils.

Biomass

Some uncertainty still exists concerning the effects of lake acidification on phytoplankton biomass. Biomass is clearly controlled by the supply of TP but whether acidification affects the rate of supply is unknown. We have compared biomass of five lakes of $pH \leq 5.1$ with 21 lakes of $pH \geq 5.6$ (Harvey, 1980) and found them not significantly different. Similarly, chlorophyll concentration was not found to be related to pH in 37 oligotrophic Norwegian lakes with different pH values. No phosphorus data were presented for these lakes (Raddum, et al. 1980).

The variability in physical and chemical features among lakes located in the same general area can be rather large and confounds attempts to compare lakes statistically. For example the Canadian Shield lakes listed by Harvey (1980) show a difference in TP concentrations among non-acidic lakes of from 2 to 16 $\mu eq\ l^{-1}$.

If phytoplankton biomass were reduced by acidification then one would anticipate increased water transparency. Increased lake clarity has occurred in a few lakes concomitant with increasing acidity (Schofield, 1973; Almer, et al. 1978) and many acidic clearwater lakes are very transparent. Almer, et al. (1978) note that humic substances are readily precipitated in the presence of Al in the pH range 4.0 to 5.0. It is their view that humic substances are bound either in the forest soils, thus reducing their input to the lake, or they are precipitated to the lake sediments. Kwiatkowsky and Roff (1976), in contrast to Raddum, et al. (1980), found water transparency and chlorophyll concentrations to inversely correlate with pH in six lakes (pH range 4.05 to 7.15) 51 kilometers south of Sudbury. The lower chlorophyll levels in the acidic lakes most probably correlate with lower concentrations of TP. The lower TP levels in the acidic lakes could indicate the inherently low rates of export of phosphorus from watershed soils of low buffering capacity; potentially, acidification of watershed soils may reduce these export rates.

BENTHIC ALGAE

In southern Norway, where precipitation is strongly acidic and many lakes and streams have been acidified, dense growths of filamentous algae and mosses have been observed in streams and, in some cases, lakes (Hendrey, et al. 1976; Hendrey and Vertucci, 1980; Lazarek, 1980).

Acidification of artificial stream channels using water diverted from Ramse Brook in Tovdal, southern Norway (Hendrey, 1976), resulted in heavy accumulation of periphytic algae, especially *Mougeotia* spp. and *Binuclearia tatrana* (chlorophyceae), *Tabellaria flocculosa* and *Eunotia lunaris* (Bacillariophyceae). In another study, Norris Brook in the Hubbard Brook watershed (New Hampshire) was acidified experimentally. This also significantly increased periphytic algae (Hall, et al. 1980). In both of these stream experiments the productivity per unit of biomass (chlorophyll *a*) decreased with acidification.

In an experimental acidification of Lake 223 waters in 10-meter diameter polyethelene tubes, Muller (1980) used H_2SO_4 to obtain treatment levels of approximately pH 6, 5, and 4, in addition to a control tube (no addition of acid) at pH 6.5. No trend was seen in biomass or productivity relative to pH but community changes did occur. The Chlorophyta, particularly *Mougeotia* spp., dominated at low pH. Diatoms, (*Achnanthes minutissima*) and *Mougeotia* spp. dominated at higher pH. Both community diversity and similarity decreased with pH. Both Hendrey (1976) and Muller (1980) observed carbon uptake by periphyton which had been removed from their substrates and incubated *in vitro*. In the artificial stream channels the rate of photosynthesis, replicated in three separate experiments, increased with decreasing pH due to the larger biomass at lower pH, but the photosynthesis per unit biomass (P/B) decreased with pH (Hendrey, 1976). While there were obvious differences in the tolerance of the algal species to low pH, and this must certainly enter into community composition (Hendrey, 1976), it cannot explain the increased biomass at low pH. For example, *Tabellaria flocculosa* which dominated acidified stream communities in three of five replications of the Tovdal experiment, has been found to have a pH optimum between 5.0 and 5.3 (Cholonky, 1968) or higher (Kallqvist, et al. 1975). The niche breadth with respect to pH for this species may be wider than that of others but its optimum is not at pH 4. Therefore, some explanation other than tolerance (Muller, 1980) must be found for observed increases in algal abundance in streams of low pH. We will discuss this later in the paper.

MACROPHYTES

Evidence of changes in macrophytic community structure following acidification has been obtained for a few lakes. Lake Colden, N.Y., which has been acidified by ca. 97 micro equivalents per liter and now has a pH near 4.9, was surveyed in 1932 and again in 1979 (Hendrey and Vertucci, 1980). Very dense stands of benthic macrophytes are frequently observed in acidic clearwater lakes. *Sphagnum* abundance was found to increase with decreasing pH in five Swedish lakes, and to have greatly expanded in acidic Lake Orvattnet (pH ca. 4.6) (Grahn, 1976). *Sphagnum* replaced more common species such as *Lobelia*, *Litorella*, and *Isoetes*. Similar density of *Sphagnum pylaesii* was observed in Lake Colden with about 300 g dry weight m^{-2}. Although *Sphagnum* is a normal component of the submerged flora of many oligotrophic softwater lakes, the extent of these stands in the Swedish lakes and Lake Colden appears to be exceptional even for acidic lakes of this type. Grahn (1976) notes that it has a high ion exchange capacity when both alive and dead (peat), and that the dense mat inhibits cycling of materials between the mineralized sediment and overlying water. Thus, *Sphagnum* may contribute to reduced plant nutrient availability in lakes where it is abundant. *Utricularia* also forms very dense stands covering large areas in Lake Colden and in Woods Lake (pH ca. 4.9), and may contribute to reducing exchange with sediments.

Macrophyte biomass increased in acidic Clearwater Lake, Ontario (Table 2). The total biomass of primary producers thus may be increased by acidification. Because of the increased clarity of acidified lakes, sediments at great depths may be available for colonization. Data on the productivity of the benthic zone in acidified lakes are not available.

Table 2. — Macrophyte biomass (dryweight), lake pH and macrophyte phytoplankton biomass ratios for three Canadian Shield Lakes in 1978 (unpubl. data provided by I. Wilde and G. Miller, Ontario Minist. Environ.)

			Biomass Ratios	
	Lake pH	Macrophyte Biomass (g m^{-2} of vegetated zone)	Macrophyte / Phytoplankton	Macrophyte Shoots / Phytoplankton
Harp	6.8	66	6.1	2.2
Red Chalk	6.6	57	4.9	2.1
Clearwater	4.4	240	73.0	19.7

Table 3. — Comparison of standing stocks of benthic invertebrates from oligotrophic lakes of different pH's in Northern Ontario.

Lake	A_o(ha)	Z_{max}	pH	Abundance of Benthos numbers m^{-2}	Source
Middle	28	15	4.4	650	Scheider, et al. 1975, 1976a
Hannah	27	9	4.3	1,200	"
Lohi	41	20	4.4	1,100	"
Clearwater	77	22	4.3	1,000 - 4000	"
Lumsden	22	23	4.6	6,000 - 17,000	Beamish, 1974
George	148	30	5.2	3,600 - 18,800	"
Nelson	309	50	5.7	1,100 - 2400	Scheider, et al. 1976b
Lake Type D	12 - 44	13	>5.7	500 - 1600	Hamilton, 1971

ZOOPLANKTON

From regional surveys conducted in Canada and in Scandinavia it may be concluded that acidification of lakes is accompanied by a two to threefold reduction in richness of species of crustacean zooplankton (Almer, et al. 1974; Hendrey and Wright, 1976; Sprules, 1975). Although data are less abundant it appears that rotifer community diversites are also reduced (Almer, et al. 1974; Roff and Kwiatkowski, 1977); some rotifer species, however, are exceptionally tolerant of acidic environments (Smith and Frey, 1971).

Changes in community structure are most noticeable at pH<5.0 . In Ontario, two of the species most commonly observed and numerically dominant in non-acidic lakes, *Diaptomus minutus* and *Bosmina longirostris*, become even more important in acidic lakes as other less tolerant species such as *Daphnia* decline (Sprules, 1975). As fish are usually absent at these levels of pH, the dominance of the zooplankton community by small-bodied herbivores contradicts the observation that in the absence of fish predation, the dominant zooplankton will be large-bodied (Walters and Vincent, 1973; Dodson, 1974; Brooks and Dodson, 1965). This may result simply from differences in tolerance among species to depressed pH. Physiological bases for such differences have been demonstrated (Potts and Fryer, 1979). Dominance by small-bodied herbivores also may be attributed to low levels of invertebrate predation in the absence of fish predation (Lynch, 1979). Yan and Strus (1980) and De Costa and Janicki (1978) showed, for example, that dominance by *Bosmina longirostris* in acidic lakes was most evident only after the dominant cyclopoid predator had declined in density. The domination by small-bodied herbivores may also indicate a reduced availability of food (Goulden, et al. 1978). In some acidic lakes concentrations of dissolved organic matter and hence of bacteria may be low. A large fraction of the phytoplankton is comprised of dinoflagellates, which are not preferred prey for filter feeding herbivores (Porter, 1973). These three hypotheses for changes in structure of zooplankton communities warrant investigation.

Few studies of zooplankton are sufficiently intensive to assess whether acidification reduces zooplankton standing stocks. Apparently, there are very large year to year variations. Fo example, Yan and Strus (1980) showed that the biomass averaged over the ice-free season in Clearwater Lake, an acid metal-contaminated lake near Sudbury, Ontario, could vary up to 300 percent from year to year. What data do exist, however, suggest that acidification reduces zooplankton community biomass because both size and numbers of community dominants decline (Harvey, 1980). A consequence of this reduced biomass is reduced efficiency of energy transfer from primary to secondary trophic levels (Yan and Strus, 1980). Such a phenomenon has previously been suggested to occur in lakes acidified by mine drainage (Smith and Frey, 1971) and in acidified streams (Hendrey, 1976; Hall, et al. 1980).

BENTHIC INVERTEBRATES

Synoptic and intensive studies of lakes and streams have demonstrated that numbers of species of benthic invertebrates are reduced along a gradient of decreasing pH (Sutcliffe and Carrick, 1973; Conroy, et al. 1976; Hendrey and Wright, 1976; Borgstrom, et al. 1976; Almer, et al. 1978). The more commonly observed invertebrates in acidified waters belong to the Notonectidae (backswimmers), Corixidae (waterboatmen), Gerridae (waterstriders), Chironomidae (midges), and Megaloptera (alderfly). These may be abundant, especially in waters where fish predation has been eliminated. Trichoptera, Ephemeroptera, and Plecoptera have many species which are intolerant of low pH.

In laboratory studies, Bell (1971), Bell and Nebeker (1969), and Raddum (1978) have measured the tolerance of some invertebrates to depressed pH. Tolerance seems to be in the order caddisflies > stoneflies > mayflies. Raddum concluded that while many stonefly species did not seem to be affected by low pH, *Amphinemura sulcicollis*, *Brachyptera risi*, and *Leuctra hippopus* exhibited increased death rates and decreased caloric content when exposed to acidic water. The mayfly *Baetis rhodani*, the dominant mayfly in non-acidic Norwegian streams, did not survive experimental acidification. Roff and Kwiatkowski (1977) concluded that the diversity of benthic fauna from the La Cloche Mountain lakes near Sudbury, Ontario was reduced in two lakes of pH>5.0

While data on species occurrence are scanty, quantitative data on biomass or abundance of benthic invertebrates in acidic lakes are even less available. Raddum (1978) studied six acidic and eight less acidic lakes in Norway, all with similar substrate. He found that densities and biomasses of benthic invertebrates were reduced in the acidic lakes, but the most common animal group was the chironomids. A summary of available but scanty information from acidic and non-acidic shield lakes in Ontario suggests, in contrast, that abundance may not be reduced by depressed pH (Table 3).

Okland (1969) surveyed snail population in 832 lakes in Norway and found no snails in lakes of pH< 5.2. No comparable North American data are available. Following the observations of K. A. Okland (1969), Borgstrom and Hendrey (1976) found that the amphipod *Gammarus lacustris* adults and the tadpole shrimp *Lepidurus arcticies* could not tolerate pH< 6.0. Okland (1980) indicates that the isopod *Asellus aquaticus* may be restricted to lakes of pH>5 in Norway.

While these invertebrates are restricted to some extent by acidification, it appears that air-breathing aquatic insects, especially predators, are very tolerant of acidic environments. Population densities of Coleoptera, Corixidae, and Megaloptera increased in acidic lakes, and in the most acid lakes, Odonata species were more abundant (Raddum, 1978, 1980). No studies on changes in populations of these larger invertebrates are yet available from North America.

Following experimental acidification of Norris Brook (Hall, et al. 1980) to pH 4 in March 1977 the downstream drift of insect larvae increased 13-fold. Organisms in the collector and scraper functional groups were affected more than predators. There was also a 37 percent reduction in insect emergence with members of the collector group most affected. Invertebrates taken out of the bottom samples were reduced by 75 percent compared to the central zone, while the reduction of invertebrates in debris accumulations was 84 percent.

Low pH also appeared to prevent permanent colonization by a number of invertebrate species, primarily herbivores, of the acidified reaches of River Dudden, England (Sutcliffe and Carrick, 1973). Ephemeroptera, Trichoptera, *Ancylus* (Gastropoda), and *Gammarus* were absent. Observations that herbivorous invertebrates are especially reduced in acidified streams, as reported in Norris Brook and River Dudden, support our earlier discussion (Hendrey, 1976; Hall, et al. 1980) that this may contribute to increased algal accumulations seen in Norwegian streams, the artificial acidification at Tovdal and Norris Brook, and accumulations of benthic algae in acidic lakes.

Petersen (1980) investigated the processing of coarse particulate organic material in leaf packs in streams at different acidities. The "shredder" functional group is apparently reduced in the acidic stream. Traaen (pers. comm.; Leivestad, et al. 1976) conducted similar experiments with litter bags in Norwegian waters with differing pH. Invertebrates appeared to make a greater contribution to accelerate decomposition in less acidic waters. These studies and those in Norris Brook and River Dudden indicate that shredders, collectors, and scrapers are reduced to a greater extent than are the predatory invertebrates.

Water Hardness

The great importance of water hardness in regulating distributions of invertebrate species and their ranges of tolerance to acidity has been demonstrated, most recently by studies in Norway. K. A. Okland and Kuiper (1980) found the number of species of Sphaeriidae (small mussels) increases rapidly with increasing Ca concentration, up to 2 mg l^{-1} hardness (as CaO) (this includes over half of the 1,320 Norwegian freshwaters included in the report); the number of species also increases with pH over the pH range 4.6 to 6.9. Half of the 20 Sphaeriidae species were classified with respect to both pH and hardness requirements. Gastropods may be present at water hardness values as low as 1.5 mg l^{-1} but only if pH is 6. At greater hardness values (>6 mg l^{-1}) gastropods were found in Norwegian lakes at pH as low as 5.2 (J. Okland, 1980).

The fact that some species are more abundant in acidic softwater lakes does not necessarily imply they prefer such conditions, but may be due to the elimination of competitors who are intolerant of such conditions. This may be the case, for example, with *Asellus* aquaticus, as discussed by K. A. Okland (1980). This species is found in Norwegian waters with hardness ranging from 2.3 to 208 mg l^{-1} (as CaO) and pH 4.8 to 8.8; it decreases in frequency in lakes with hardness ≥20 mg l^{-1}, and is prominent in acidic Swedish lakes (Almer, et al. 1974). *Gammarus lacustris*, which is intolerant of low pH (Okland, 1969; Borgstrom and Hendrey, 1976), has a distribution limited to pH 6.0 and hardness > 4.5 mg l^{-} (K. A. Okland, 1980). K. A. Okland notes these two species are ecologically very similar and have a nearly allopatric distribution and, under favorable conditions, *Gammarus* apparently replaces *Asellus* by competitive exclusion, thus leaving the low pH, softwaters to *Asellus*.

Borgstrom and Hendrey (1976) found low pH inhibited moulting progression of *Lepidurus arcticus* and suggested this might be due to interference of calcium uptake (see also Sutcliffe and Carrick, 1973). Malley (1980) studied the crayfish *Orconectes virilis* (Hagen) taken from lake 223 and placed in aquaria maintained at pH 3.0, 4.0, 5.0, and 6.0. Mortality (ca. 14 percent) at pH 3 and 4 occurred during molting. Progression through molt stages was slowed by pH 5.0 and uptake of Ca^{++} was greatly retarded. Some species of crayfish do occur in rather acid waters. *Cambarus bartoni*, for example, is common in Clearwater Lake (pH 4.2).

DISCUSSION

Each species has a unique set of environmental factors at which its growth is optimized. For algae, nutrient (P, N, C, S, Fe, etc.) availability, light intensity, and temperature are the primary factors that must be optimized. Secondary factors, however, including grazing and microbial activity (which lead to nutrient regeneration and increased light penetration), and low concentrations of toxic substances such as heavy metals, are also important in determining species optima. Lake and watershed acidification alters all of these factors and increases the H^+ concentration by orders of magnitude.

Given these complexities it is not surprising that the differences in the structure of some biotic communities (e.g., phytoplankton) among acidic lakes are not yet explicable. They may in fact never be explicable. Schindler (1975) commented, for example, that despite the very precise knowledge concerning nutrient input rates, lake physics, and chemistry available for ELA lakes in northwestern Ontario, the ecosystems of the lakes were too complex to model responses to nutrient additions incorporating taxonomic detail.

Complex interactions among various trophic levels might contribute to some of the phenomena we have discussed for acid lakes. One of the most obvious features of lake acidification is that all fish have been eliminated from many such lakes. The effects of fish removal from non-acidic, oligotrophic Emmaline Lake, Colo., were studied by Walters and Vincent (1973). The zooplankton community was markedly altered, with a shift to dominance by large species, especially *Daphnia middendorffiana*; midge larvae populations increased and benthic invertebrates became dominated by large forms. In contrast, evidence from the Canadian Shield lakes indicates that smaller forms predominate in acidic lakes.

Periphyton standing crops increased greatly in the 2 years following fish removal from Emmaline Lake; this may have contributed to a "bloom" of small herbivorous midges. Periphyton was greatly reduced in the following 2 years by grazing. Henrikson, et al. (1980) note that the disappearance of fish from acidified lakes leads to a rapid increase in abundance of several species susceptible to fish predation. Vertebrate predators are replaced by invertebrate predators, notably Corixidae, *Chaoborus*, Dyticisdae, and Odonata.

Studies which link various observations, e.g., species reduction, to acidification on the basis of correlation analyses, have been criticized by Nilssen (1930) as ignoring recent analytical advances, particularly in evolutionary biology. Nilssen states that "In research on acidification the various investigated parameters are frequently plotted against ambient pH, often based on one sampling date only." The implication seems to be that some investigators have tried to correlate a parameter against one pH measurement. This, of course, is not so. What has been frequently done in regional studies is to make correlations between a parameter and pH values collected once from each of *many* lakes. This technique has been validly used for both biological and chemical analyses in the Swedish and Norwegian synoptic surveys. Nilssen is correct in pointing out that correlation, e.g., decreased species number with decreased pH, does not prove direct cause. Decreased species number may prove to relate more directly to increased Al concentrations and/or removal of predators than to the concentration of H^+ The driving variable in the changes we have discussed in this review is increased H^+ loading to lakes, nonetheless.

Changes in functional guilds of organisms will undoubtedly have affects at other trophic levels. We present the following hypothetical linking of biological observations in acidic lakes. The removal of fish predation increases predatory invertebrate abundance. This results in increased invertebrate predation on collectors, shredders, and scrapers and reduces the activities of these guilds. Accumulation of attached algae and benthic litter would thus be enhanced by these changes. Benthic plants, especially algae and litter, are in fact abundant in many acidic lakes and streams. This hypothetical sequence is composed of elements in a chain that have not yet been conclusively linked. Quantitative studies of transfers of mass or energy between trophic levels in acidified lakes are lacking. This is the type of research which could now be most fruitful to provide an integrated understanding of acidification impacts on aquatic flora and fauna.

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RESPONSES OF FISHES TO ACIDIFICATION OF STREAMS AND LAKES IN EASTERN NORTH AMERICA

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ABSTRACT

Precipitation in eastern North America is acidic and contains elevated levels of heavy metals. As a result of this precipitation the pH of lakes and streams has declined and metal concentrations have increased in several areas. Episodic decreases in pH and increases in metal concentrations are associated with spring snowmelt and heavy rains. These changes in water quality adversely affect resident fish populations, reducing growth rate, increasing frequency of skeletal deformities, and eliminating sensitive species through mortality or reproductive failure. Small-scale remedial action has been taken in some areas, including lake neutralization, hatchery stocking, and selective breeding for acid tolerance. In spite of all of the studies and observations, the causes of declining fish populations in acidified waters have not yet been identified. Classical field observations and laboratory bioassays do not provide enough information to demonstrate the effects of acid precipitation on fish populations. Innovative experiments will be required to provide definitive answers to these questions.

INTRODUCTION

In eastern North America precipitation is now more acidic than in Scandinavia, where acid rain problems have been documented for a number of years. The median annual pH for 1978-79 ranged from 4.0 to 4.4 (Gibson and Baker, 1979; Atmos. Environ. Serv., 1979). Following an examination of historical data, Cogbill (1976) concluded that precipitation during the 1920's was low in acidity (probably pH 5.5). Precipitation had become more acidic than normal by 1955, and has increased steadily since then. This decline in pH has been attributed to increased NO_x emissions, and to a variety of factors that have increased long-range transport of SO^2 emissions, such as taller smokestacks and more use of particle precipitators in smokestacks (Likens and Bormann, 1974). Studies of the magnitude and distribution of SO^2 and NO_x in eastern North America have shown that in both cases the highest emissions are located [illegible] the industrial Midwest and Great Lakes areas of the United States and Canada (Altshuller and McBean, 1979). Prevailing wind directions and storm tracks then carry these emissions to the northeast (Altshuller and McBean, 1979; Schlesinger, Reiners, and Knopman, 1974).

The metal content of acidic precipitation is higher than normal precipitation (Delisle, Kloppenburg, and Sylvain, 1979; Elgmork, Hagen, and Langeland, 1973; Lazrus, Lorange, and Lodge, 1970; Ruhling and Tyler, 1973; Schlesinger, Reiners, and Knopman, 1974). Metals which have been found at elevated levels include lead, zinc, copper, iron, manganese, nickel, mercury, and cadmium. They probably come from fossil fuel combustion and metal smelting (Likens and Bormann, 1974).

The effect of acidic precipitation on an aquatic system is determined by the geochemistry, geomorphology, and hydrodynamics of the watershed. These factors determine the capacity of soil and water to neutralize acids and resist pH change in surface waters. In watersheds where the resulting buffering capacity is low, the pH of surface waters has decreased. Such changes have been recorded in Nova Scotia, Ontario, New York, New Jersey, New Hampshire, and Maine (Watt, Scott, and Ray, 1979; Beamish and Harvey, 1972; Schofield, 1976; Johnson, 1979; Hendrey, et al. 1980; Davis, et al. 1978). Trace metal levels in acidified lakes are higher than in comparable lakes located in areas receiving normal precipitation (Wright and Gjessing, 1976; Beamish and Van Loon, 1977). These elevated metal levels may result from direct input from precipitation or leaching from soils or sediments by hydrogen ions in the precipitation. The latter mechanism is especially important for aluminum and manganese (Cronan and Schofield, 1979; Henriksen and Wright, 1977).

Contaminants brought to aquatic systems by precipitation vary seasonally and with individual storm events. In eastern North America the greatest input occurs with snowmelt, when pollutants stored in the snowpack are released over a short period of time (Schofield, 1977; Jeffries, Cox, and Dillon, 1977).

OBSERVED IMPACTS ON FISH

Long-term and episodic changes in pH and metal content of surface waters have affected the biota inhabiting these waters. Effects on fish have been widely reported, probably because fish are highly sensitive to acid and heavy metals and are one of the most visible components of aquatic ecosystems. The observed effects include direct mortality, reproductive failure, reduced growth, and skeletal deformities.

The disappearance of various fish species from acidified lakes has been recorded. In a study of 68 Ontario lakes the number of species of fish present was found to decrease as measured lake pH decreased (Harvey, 1975). In North America, there are many recorded instances of fish disappearing from lakes in Ontario and New York (Beamish, et al. 1975; Schofield, 1976). The apparent pH at which they disappeared is listed in Table 1. Either these fish died or they failed to reproduce.

Table 1. — Field studies on effect of lake acidification on natural fish pond.

Family and Species	pH at which population ceases reproduction, declines or disppears
Salmonidae	
Lake trout (*Salvelinus namaycush*)	5.2-5.5 (1) 5.2-5.8 (2)
Brook trout (*Salvelinus fontinalis*)	4.5-4.8 (3) - 5 (7)
Aurora trout (*S. fontinalis timagamiensis*)	5.0-5.5(4)
Arctic char (*Salvelinus alpinus*)	- 5 (5)
Rainbow trout (*Salmo gairdner*)	5.5-6.0 (3)
Brown trout (*Salmo trutta*)	5.0 (3) 5.0-5.5 (6) 4.5-5.5 (8)
Atlantic salmon (*Salmo salar*)	5.0-5.5 (3)
Lake herring (*Coregonus artedii*)	4.5-4.7 (1) < 4.7 (2)
Esocidae	
Northern pike (*Esox lucius*)	4.7-5.2 (2)
Cyprinidae	
Lake chub (*Coueslus plumbeus*)	4.5-4.7 (1)
Roach (*Leuciscus rutilus*)	5.3-5.7 (5) 5.3-5.7 (5)
Catostomidae	
White sucker (*Catostomus commersoni*)	4.7-5.2 (1) (2)
Ictaluridae	
Brown bullhead (*Ictalurus nebulosus*)	4.5-5.2 (1) (2)
Percopsidae	
Troutperch (*Percopsis omiscomaycus*)	5.2-5.5 (1)
Gadidae	
Burbot (*Lota lota*)	5.5-6.0 (1) 5.2-5.8 (2)
Centrarchidae	
Smallmouth bass (*Micropterus dolomieul*)	5.5-6.0 (1) > 5.5 (2) - 5.8 (9)
Rock bass (*Ambloplites ruperstris*)	4.7-5.2 (1) (2)
Pumpkinseed sunfish (*Lepomis gibbosus*)	4.7-5.2 (1) (2)
Percidae	
Walleye (*Stizostedion v. vitreum*)	5.5-6.0 (1) 5.2-5.8 (2)
Yellow perch (*Perca flavescens*)	4.5-4.8 (1) < 4.7 (2)
European perch (*Perca fluviatilis*)	5.0-5.5 (10)

References:
(1) Beamish, 1976; (2) Beamish, et al. 1975; (3) Grande, Muniz, and Anderson, 1978; (4) Anonymous, 1978; (5) Almer, et al. 1974; (6) Jensen and Snekvik, 1972; (7) Schofield, 1976; (8) Wright and Snekvik, 1978; (9) N.Y. State Dep. Envron. Conserv. 1978, (10) Runn, Johansson and Milbrink, 1977.

Acute mortalities of fish which may result from acidification and/or metal toxicity are rarely observed under field conditions, and mortalities during episodic inputs of hydrogen ion may be more common than has been commonly observed. Mortality of Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) have been recorded in Norway, usually following a sudden spring melt or heavy autumn rain (Jensen and Snekvik, 1972; Leivestad and Muniz, 1976; Leivestad, et al 1976). The pH measured during these mortalities ranged from 3.9 to 4.6. Mortality of brook trout (*Salvelinus fontinalis*) occurred during spring snowmelt in a New York laboratory which was rearing fish in water piped from a nearby stream (Schofield,1977). The lowest pH was about 5.2 and aluminum concentration reached 1 mg/l. Mortality of Atlantic salmon fry has been observed in hatchery pools fed water with a pH of 5.0 from the Mersey River, Nova Scotia (Farmer, et al. 1980). Embryos and fry are generally more sensitive to acid and metals than adult fish (McKim, 1977), and mortality of these stages would be difficult to observe in nature.

Reproductive failure has been reported in fish inhabiting acidified lakes, resulting in their gradual disappearance over a period of several years. Missing age classes was first observed, then all young age classes were absent, and finally the populations consisted only of a few large, old fish (Beamish, et al 1975; Beamish, 1976; Ryan and Harvey, 1980). This apparent reproductive failure could have resulted from either failure of fish to reproduce and deposit the normal number of viable eggs, or post-spawning mortality of embryos, larvae, or fry. Failure to deposit eggs were observed in populations of white sucker (*Catostomus commersoni*) from acidified lakes in Ontario (Beamish, 1976; Lockhart and Lutz, 1976). Post-spawning mortalities of early life stages of Atlantic salmon have been produced in laboratory studies under conditions similar to those found in acidified lakes and streams (Daye and Garside, 1977, 1979).

Reduced growth rates have been reported for white suckers and yellow perch (*Perca flavescens*) from acidified lakes (Beamish, 1974b; Ryan and Harvey, 1980). Conversely, increased growth under similar conditions was observed for older yellow perch and rock bass (*Ambloplites rupestris*) (Ryan and Harvey, 1977, 1980). Skeletal deformities have been reported for white sucker (Beamish, 1974a).

CAUSES OF IMPACTS ON FISH

Laboratory studies of the effects of reduced pH on fish are summarized in Table 2. These studies show that, depending on species, adult and embryo life stages are least sensitive to pH, while production of viable eggs, egg hatchability, fry mortality or growth are the most sensitive biological parameters. Laboratory mortality data should be interpreted with caution. Acidification of high pH water without adequate aeration will produce high free CO_2 concentrations,

Table 2. — Reduced pH levels found in laboratory experiments to cause various adverse effects on several fish species. Duration of exposure varied from 4 days to life cycle.

Family and Species	Reduced v Viable Eggs	Reduced Egg Hatchability	Increased Embryo Mortality	Increased Fry Mortality	Increased Adult Mortality	Reduced Growth	Ceased Feeding	Tissue Damage	Bone Deformity
Salmonidae									
Brook trout (*Salvelinus fontinalis*)	5.0(1)	6.5(1) 5.6(12)	4.5(9)	6.1(1) 3.5(17)	4.1(3)	6.5(1)			
Arctic char (*Salvelinus alpinus*)						4.6(19) 4.8(20)		5.2(16)	
Rainbow trout (*Salmo gairdneri*)			5.5(10) 4.3(18)	4.1(10) 4.8(20)					
Brown trout (*Salmo trutta*)		4.0(2)	< 5.0(9)						
Atlantic salmon (*Salmo salar*)		4.0(2) 4.0-5.5(15)	3.6(5) 3.9(6) 5.0(9) 3.4-4.4(18)	4.0(5) 4.3(6) 4.3(18)					
Esocidae									
Northern pike (*Esox lucius*)			5.0(8)						
Cyprinidae									
Roach (*Leuciscus rutilus*)		5.6(7)							
Fathead minnow (*Pimephales promelas*)	6.6(14)	5.9(14)		5.9(14)	2.1(14)	4.5(14)			
Catostomidae									
White sucker (*Catostomus commersoni*)		4.5(13)		5.3(13) 4.0(4)		4.5(4)	4.5(4)		4.2(4) 5.0(12)
Percidae									
European perch (*Perca fluviatilis*)		5.6(7)							

References:

(1) Mendenez, 1976; (2) Carrick, 1979; (3) Robinson, et al. 1976; (4) Beamish, 1972; (5) Daye and Garside, 1977; (6) Daye and Garside, 1979; (7) Johansson and Milbrink, 1976; (8) Johansson and Kihlstrom, 1975; (9) Johansson, Runn, and Milbrink, 1977; (10) Kwain, 1975; (11) Runn, Johansson, and Milbrink, 1977; (12) Trojnar, 1977b; (13) Trojnar, 1977a; (14) Mount, 1973; (15) Peterson, Daye, and Metcalfe, 1980; (16) Daye and Garside, 1976; (17) Daye and Garside, 1975; (18) Daye, 1980; (19) Lievestad, et al. 1976; (20) Edwards and Hjeldnes, 1977.

which interfere with the mortality effects of hydrogen ion. These data may therefore report mortality at higher pH than would occur in the absence of free CO_2.

Chronic exposure to reduced pH is unlikely to kill adult fish in acidified lakes. Lakes rarely have pH below 4.5 even in the most extreme cases; this probably will not be toxic to adult fish. However, a sudden reduction in pH can cause mortality at a much higher pH than chronic exposure to gradually reduced pH. Episodic pH changes in early spring can cause observed fish mortalities. Death will also occur at higher pH in water with very low ionic content than in water with moderate or high ionic content, because of increased osmotic stress (Fromm, 1980).

Mortality of fish at low pH has been attributed to failure of ion regulation or to asphyxiation. Fish collected from the Tovadal River, Norway, during an acid-caused fish-kill had lower plasma sodium and chloride levels than fish from unaffected sections of the river. The reduced levels were comparable to those found in fish killed by low pH in a tank experiment (Leivestad and Muniz, 1976). Transfer from pH 7 to pH 4 caused a threefold increase in sodium loss from brown trout, sufficient to cause death in 24 to 48 hours (Potts, 1979; McWilliams and Potts, 1978). The loss of ions appeared to result from an increased efflux across the gills, rather than a reduced influx (Packer and Dunson, 1970; McWilliams and Potts, 1978; Fromm, 1980). Exposure to increases in hydrogen ion concentrations increases gill membrane permeability. Hydrogen ions from the environment diffuse in and sodium and other ions from the blood diffuse out. Gill membrane permeability is mediated by calcium ion, and possibly other divalent cations, with the presence of calcium reducing permeability. Thus, with increasing calcium levels, the loss of ions is reduced and the lethal pH decreases (Fromm, 1980).

Low pH may interfere with respiration through several different mechanisms. Exposure to elevated hydrogen ion may cause excessive secretion of mucus from the gills, thereby reducing the rate of oxygen diffusion across the gill surface (Daye and Garside, 1976; Dively, et al. 1977). The increased influx of hydrogen ion reduces blood pH, which in turn reduces the oxygen-carrying capacity of hemoglobin. Fish respond to chronic acid exposure by increasing the hematocrit index, hemoglobin content of blood, and hemopoietic activity to maintain oxygen carrying capacity (Fromm, 1980).

It is not possible currently to determine whether osmotic or respiratory effects, or both are responsible for death of fish at low pH, or whether some other factor may be involved. Fromm (1980) speculated that reduced pH may destroy enzyme activity; however, he also noted that trout are remarkably tolerant of changes in blood pH per se, which would indicate that enzyme activity was not affected. The action of reduced pH to increase the toxic effects of metals, such as aluminum, may be more important than the direct effects of pH.

The failure of fish exposed to low pH to produce viable eggs has been explained by changes in serum calcium levels in females. Beamish (1976) and Lockhart and Lutz (1976) observed that the failure of female white suckers exposed to low pH to produce viable ova was coincidental with lower than normal serum calcium levels. Serum calcium, in the form of complex calcium phospho-proteinate, normally increases in females during the period of ova development. Ruby, et al. (1977) showed that oogenesis in flagfish (*Jordanella floridae*) was reduced by exposure to reduced pH because protein production was disturbed, leading to improper yolk formation. Calcium is important in the transfer of protein to the yolk.

Reduced egg hatchability at low pH in some fish species apparently results from failure of the choriolytic enzyme to properly degrade the chorion. This was observed in European perch (*Perca fluviatilis*) by Runn, Johansson, and Millbrink (1977) and in Atlantic salmon by Peterson, Daye, and Metcalfe (1980). This supports Fromm's (1980) speculation concerning the effect of reduced pH on enzyme activity.

The effect of reduced pH on growth is ambiguous. Growth of some fish may either increase or decrease following acidification. Increased growth of fish which survive acidification may result from decreased competition for food following the disappearance of acid sensitive competitors (Ryan and Harvey, 1977, 1980). Conversely, the growth rate of fish may decline in the face of abundant food, which is explained as a sublethal stress response to the increased acid (Beamish, 1974b). One laboratory study produced reduced growth in brook trout exposed to pH 4.6 (Leivestad, et al. 1976), but only during the first 3 months of exposure in another (Menendez, 1976), On the other hand, reduced pH did not affect growth in brown trout (Jacobson, 1977).

Skeletal deformities observed in natural populations of white sucker (Beamish, et al. 1975) were also produced by reduced pH in laboratory bioassays (Beamish, 1972; Trojnar, 1977a). This may be caused by loss of serum calcium under acid stress. However, the effect has not been observed in other species.

In addition to reduced pH, metal concentration, either alone or in conjunction with reduced pH, was responsible for fish losses in many acidified lakes. Cronan and Schofield (1979) and Schofield and Trojnar (1980) showed that mortality of brook trout in New York was caused by aluminum and pH in combination, rather than by either factor singly. Similar results were reported by Grahn (1980) for brook trout and ciscoe (*Coregonus albula*), by Hermann and Baron (1980) for brook trout, and by Muniz (1980) for brown trout.

The toxicity of aluminum varies with pH and the presence of complexing agents. The most toxic forms of aluminum ($Al(OH)^{++}$, $Al(OH)_2^{+}$, $Al(OH)_4^{-}$) are present at ph 5, and the toxicity declines at both higher and lower pH. Aluminum complexed with organic matter is not toxic to organisms (Driscoll, et al. 1980). At pH 5, aluminum concentrations of 0.2 mg/l or greater cause gill hyperplasia and mucus secretion (Schofield and Trojnar, 1980). These pH and aluminum levels are common in acidified waters (Wright and Henriksen, 1978; Dickson, 1975; Schofield, 1977).

Yan, Girard, and Lafrance (1979) reported mortality of rainbow trout (*Salmo gairdneri*) stocked in an acidified lake which had been chemically neutralized. Fish had been eliminated from the lake by acidification. The mortality was attributed to copper or copper and zinc in combination. Copper concentrations were 42 to 67 ug/l and zinc concentrations were 23 to 33 ug/l. Copper concentrations as low as 9.5 mg/l were reported to be toxic to brook trout embryos and juveniles in laboratory exposures (McKim and Benoit, 1971). Copper concentrations up to 450 mg/l have been measured in lakes near Sudbury, Ontario (Adamski and Michalski, 1975). The toxicity of many metals increases as pH and calcium decline (Chrost and Pinko, 1980; Franzin and McFarlane, 1980.)

REMEDIAL ACTION

Action has been taken to counter the effects of acid precipitation on selected fish populations. A variety of approaches is now under study, including lake neutralization, hatchery stocking, and selective breeding for acid tolerance. Lake neutralization is the most widely used approach. Lakes are treated with $CaCO_3$, $Ca(OH)_2$, or both. The combination treatment appears to have given the most satisfactory results to date (Scheider and Dillon, 1976). In New York, lakes totalling 819 acres are now being treated with lime; viable brook trout fisheries survive (Pfeiffer, pers. comm.). In Ontario several lakes have been neutralized, or neutralized and fertilized, and changes in biota monitored (Scheider and Dillon, 1976; Dillon, et al. 1977). However, rainbow trout stocked in one such treated lake survived only a few days, apparently as the result of copper toxicity (Yan, Girard, and Lafrance, 1979). Metal concentrations in this lake had been elevated by acidification and atmospheric input from metal smelters. Apparently neutralization alone was not sufficient to reduce metal toxicity.

Stocking hatchery fish at the fingerling stage avoids exposure of sensitive early life history stages to environmental stress and maintains fish populations in lakes in New Hampshire and Maine, that would otherwise not support fish (Haines, unpubl. data). Gunn (1980) reports that *in situ* incubation of eyed eggs of fish (species not stated) in spawning boxes filled with limestone is a promising technique for maintaining fish populations in acidified lakes. The limestone protects embryos and larvae from toxic acids or metals in lake water.

Various populations of brook and brown trout have been shown to differ in tolerance to acid, and this tolerance is heritable (Robinson, et al. 1976; Edwards and Gjedrem, 1979). Thus a selective breeding program to improve fish survival in acidified waters is possible. This technique has been successfully employed in New York (Pfeiffer, pers. commun.).

DETERMINATION OF THE MECHANISMS OF ACID PRECIPITATION IMPACT ON FISH POPULATIONS

In spite of all of these studies, we remain ignorant of the true relationship between acidification from precipitation and loss of fish populations. Laboratory findings of pH tolerance cannot be directly related to field observations to determine the reasons for the loss of fish. Possible explanations for these discrepancies include effects of metals, singly or synergistically, and episodic changes in water quality that occur during the acidification process. It is possible that episodic changes in pH and metal content stress critical life history stages (e.g., eggs, fry, maturing adults). Descriptive field studies and classical laboratory bioassays do not provide sufficient information to demonstrate how acid precipitation affects fish populations. We believe that innovative approaches will be required to provide these answers. We advocate the following approaches:

1. Intensive case studies: application of population dynamics research to fish populations in carefully selected waters undergoing acidification. These studies would demonstrate the relative importance of adult mortality, reproductive failure, and reduced growth in determining effects of acidification on the populations.
2. Experimental manipulations of pH in field situations: induce both chronic and episodic acidification in sensitive waters not subject to acidic precipitation and determine resulting fish population responses.
3. Detailed studies of pH-metal-ligand interactions: the relationship between pH, metal form, and ligands in determining toxicity must be detailed in laboratory studies, and field sampling must include more detailed chemical analyses than in the past.

These approaches will provide insight into the acidification process which has not been obtained from previous studies.

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FUTURE TRENDS IN ACID PRECIPITATION AND POSSIBLE PROGRAMS

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ABSTRACT

Remedial action programs focus on sources, terrestrial and aquatic effects, and socio-economic aspects of acidic precipitation. One research thrust is to assess the critical height of a source emission with respect to the production of acidic aerosols; these soruces then should be given priority in an abatement program. Modeling seems to be the best approach to this problem, and field data exist for various point source emissions for various heights to calibrate the model. Abatement of aquatic effects can be effected through liming, nutrient enrichment, and modification of watershed hydrology. Mass budget calculations and field studies are required to evaluate these alternatives. Finally, abatement costs should be compared with the costs of tax reduction and direct subsidies to the user to decrease energy consumption and thus emissions. The recent U.S. National Academy of Science energy report suggests that a 50 percent per capita reduction in energy consumption is achievable. Assuming a proportional correlation between emissions and deposition, this energy saving is comparable to the billin dollars required to abate existing emission sources with NO , SO , and particulate controls.

* * *

INTRODUCTION

Any attempt to propose research and abatement programs for acid precipitation must consider a number of related phenomena:

1. Emissions resulting in acid precipitation are complex and tied to the energy demands, technological and economic conditions, and lifestyles of one or more countries.

2. Acid aerosols and acid precipitation cover a wide area, containing many millions of square kilometers (Hoffman and Rosen, 1980).

3. The impacts of aerosols and acid precipitation are multiple, negatively affecting manmade structures, health, aquatic ecosystems, and possibly forestry and agricultural productivity. The economic effects of SO_2—SO_4 have been estimated in the tens of billions of dollars (U.S. EPA, 1979).

4. There is a strong possibility that acid aerosol and precipitation impacts will become more severe in the next 20 to 30 years.

5. There will be a strong correlation between energy use and the form of energy and acid precipitation in the next 20 to 30 years.

The rationale used here to arrive at suggested research studies and programs to abate acid precipitation is (1) consideration of emissions, transport of pollutant, and impacted areas; and (2) consideration of immediate (and generally temporal) abatement, and also long-term (decades) and generally permanent abatement. Controlling emissions is considered both from technological abatement ("scrubbing") and from the reduction of energy losses ("waste"). Transport of pollutants focuses on the sensitivity of stack height to long-range transport and on chemical conversion processes. Impacted area abatement focuses directly upon treatment of aquatic watersheds ("liming").

The purpose here is to project emissions into the 21st century using energy models as a base and to suggest general approaches and specific research programs for remedial action.

FUTURE ENERGY DEMANDS AND PROGRAMS

The kind of fuel and amount of waste energy is almost directly related to acid precipitation aerosols. Therefore it is pertinent to examine various energy projections focusing on emissions. Perhaps the most profound statement regarding future energy demands and programs is that there are many uncertainties for decisionmakers. Unfortunately, the large scale of the problem demands decisions and investments now to produce payoffs 20 to 30 years hence.

Some of the most significant factors that will affect energy demand and policy are the evolving relationships between economic factors and energy consumption, conservation, and technological change. Equally significant will be the reactions of society's next generation to altered lifestyles. All of these are related in a most complex way, and can be very much affected by consumer attitude, government policies, pricing, and technological change. For example, there has been much discussion between economists relying on supply/pricing strategy and resource scientists seeing finite limits to availability of energy material (Hayes, 1979). Furthermore, there is no agreement that a stable economy requires a nearly constant "GNP/energy consumption" ratio; some suggest that this ratio may double over time with a stable GNP growth of

2 percent and "business as usual" (Natl. Res. Counc. 1980).

The large degree of flexibility and uncertainty is perhaps best summarized by the recent compilation (Marshall, 1980) showing changes made in forecasts from 1972 to 1978 by "low-growth" and "high-growth" proponents for energy requirements in the year 2000 to 2010. The 1972 "low-growth" projection of 125 quads (10^{15} BTU) equals the 1978 prediction of the "high-growth" advocates; and the overall range of forecasts in this 6-year period is sixfold (33 to 190 quads) for total energy demand. These figures can be compared to total energy input of about 78 quads in 1977.

Amid these uncertainties, there are, however, some strong future probabilities that directly relate to energy consumption and pollution emission of acidic aerosols. The following use the final report of the Committee on Nuclear and Alternate Energy Systems (CONAES) (Natl. Res. Counc. 1980) as a guide.

(1) Coal will be a significant fuel in the future, especially for thermal generating plants. Energy production using coal will about double in the next 20 to 30 years. (2) Electrical generation from thermal generating plants will become a more important form of energy, decreasing the ratio of net energy/gross energy. As an approximate and limiting assumption, the difference between gross and net energy may be assumed to be proportional to the emission load of SO_2.

Figure 1 shows the forecasts from CONAES models comparing total energy demand to various price assumptions. Figure 2 shows the coal energy demand for the same price assumptions. The important difference between these two diagrams is the comparison relative to present (1975) values. For example, total energy demand projections for 2010 are a little less than 1975 demand for case A, maximum price with conservation. But coal energy demand (assumed proportional to unabated SO_2 emissions) projection for case A is 1.6 times 1975. In short, there is not a quantitative comparison between energy demand and emissions. This is due to the increased use of coal in projection, but more important, the decreased efficiency due to the increased use of electricity.

This model suggests then that emissions of SO_2 without scrubbing will increase at least 60 percent in the 21st century. Other models need to be studied from an environmental emissions perspective in order to test this conclusion.

The distribution of coal energy demand by the consuming sector focuses on areas requiring most attention. The three sectors defined are transportation, buildings, and industry; transportation has almost no coal energy demand, and industry consumes about 62 percent of coal energy demand for all scenarios except the lowest price where it decreases.

Another CONAES scenario, "CLOP", assumes implementation of advanced technology, a strong environmental conscience, low material consumption, etc. (Natl. Res. Counc. 1980). The CLOP projections result in the same total energy consumption as does the highest price plus conservation model (A*); however, presumably (not stated by CONAES) the energy loss and especially the pollutant emissions would be lower due to increased use of solar energy.

In addition, the following summary questions regarding energy appear to be important with respect to emissions and the acid rain problem:

1. Assuming a major increase in coal use, can we learn to use coal cleanly?
2. Given that 50 percent of the fuel consumption by automobiles is made for trips of less than 5 km (Cook, 1973), can cities and transportation be restructured in the next 20 to 30 years?
3. Can cogeneration facilities become a significant function in the next 20 to 30 years?
4. Can we learn to use wastes, many of which are organic pollutants now, as an important source of fuel?
5. Can industrial energy consumption per unit of production be reduced considerably (20 to 50 percent) by waste recycling, product substitution, and technological innovation?

SOURCE EMISSION ABATEMENT

As previously mentioned, source emission abatement focuses upon coal burning plants. Abatement in this context can be achieved by technological scrubbing, by reduced energy consumption, and by use of alternate energy sources. In the following discussions, it is assumed that abatement will take place. Therefore, scrubbing costs might be replaced, for example, by costs of new technology or costs of developing alternate energy. Estimated efficiencies for removing SO_2 and associated pollutants from plants are 80 to 90 percent for new installations and 50 percent for retrofits (U.S. EPA, 1979a, b) with a cost of billions of dollars per year for treatment and disposal in the United States. These efficiencies and costs serve as a reference for alternate approaches to coal emission solutions.

Present sulfur abatement procedures for coal consist of physical cleaning, flue gas desulfurization (FGD), fluidized bed combustion (FBC), liquefaction, and gasification. In addition, chemical cleaning may become important (Ruether, 1979). Physical cleaning will remove inorganic sulfur, which accounts for about 50 percent of the total sulfur, but only 10 to 20 percent of U.S. coals can be reduced to acceptable levels by this technique. FGD can attain 90 percent removal efficiencies, given the availability of trained personnel (U.S. EPA, 1979b). FGD removal should be considered for cycled water in some fly ash lagoons, because some fly ash slurries attain a pH of 11 to 12; this pH should increase the SO_2 oxidation rate. Other fly ash slurries are acid. The characteristics of fly ash in different coals should be investigated, since using fly ash would cut chemical costs and solid waste disposal. Ponding of fly ash and water would remove the possibility that trace metals would be mobilized in surface and ground water; furthermore, the trace metals might be exploited as a resource in the waste.

Some recent studies have characterized the chemical nature of fly ash (Talbot, Anderson, and Andren, 1978; Edzwald and DePinto, 1978). During the past decade, studies and proposals have been made for using fly ash in acid mine neutralization (Tenney and

Echelberger, 1970). This requires a detailed mineralogical and chemical study of the solid along with a solution study of the aqueous slurry. Clearly much more research must focus on coal and its wastes, their abatement and resource potentials.

Many have emphasized the use of other "clean" sources of energy. Achieving abatement by using alternate sources and/or emissions involves many uncertainties as well as many possibilities. Selecting alternate sources of energy hinges on economic feasibility, amount of energy required to obtain the alternate sources, potential duration of supply, and the likelihood of developing other environmental problems. Nuclear and solar sources are often discussed; geothermal sources may be significant to specific areas. Nuclear generation is a contentious issue (e.g., Natl. Res. Counc., 1980; Holden, Smith, and Morris, 1979) and is not considered as an alternate here.

Solar energy devices are normally dismissed as requiring large investments to achieve long-term returns (Hayes, 1979; Natl. Res. Counc. 1980); only passive collectors for direct heating are thought to be feasible for immediate use. Furthermore, it is possible that, through the year 2000, the energy required to manufacture solar devices may approximate the energy produced from such devices (Whipple, 1980); presumably this energy would be from coal or nuclear fuels. High initial energy consumption would be required to develop solar energy, but over its lifetime the solar device would engender fewer emissions than would other sources; the energy payoff period would be approximately 10 years.

Other novel sources of energy such as wind, solar electric, ocean thermal, and tides are presently ruled out because their costs are estimated to be approximately 10 times that of available sources of energy (Natl. Res. Counc. 1980); however, environmental costs are not considered in these estimates.

Obviously, there is a balance between investment in industrial innovation to minimize emissions and the scrubbing costs of these emissions. It is not clear that the costs of reducing emissions either by scrubbing or by minimizing energy used have been carefully considered in developing most energy productions. It would be desirable to focus specifically on the overall abatement aspects of these projections in context to the whole, to test the effect of various mechanisms such as price, emission regulations, and alternate energy sources on the kinds and amounts of emissions. For example, according to projections, increasing energy prices fourfold will not modify the kind of energy or the amount used in the industrial sector as much as one might imagine. Moving into the real world, one wonders what investments in innovative technology in fuel supplies (i.e., solar) and in industrial technology might be brought on by such a large price change.

Energy price and conservation seem to be important factors in energy projections for buildings to the year 2000 to 2010. The CONAES models suggest energy consumption in buildings will decrease by 60 percent with a doubling of price adjusted for inflation. Presently enacted conservation programs which include standards for appliances, thermal performance for new construction, and retrofitting have been projected to decrease energy by 20 percent (Hirst and Hammon, 1979). Other important considerations re energy consumption in buildings will be architectural design and the development of district heating systems. Architectural design must minimize space per function and concentrate upon closed space rather than open space design; the latter permits the efficient use of computer controlled zone heating and cooling systems. Many projections suggest 50 percent of building energy use will be from purchased electricity. If this is so, a marked improvement in energy efficiency (about 50 percent from Swedish experiences) can be attained from implementing district heating systems in conjunction with electrical generating plants. Research and incentives toward these ends are needed.

Decreases in energy used for transportation appear to be equally sensitive to price increases and conservation efforts. A doubling of price with conservation effort would decrease energy consumption by about 50 percent. Oil is projected to be the only significant fuel in 2000 for transportation.

In summary, a doubling of price with probable conservation will markedly reduce energy consumption in buildings and transportation. It is not clear what effect other energy reducing activities, especially design, will have on energy consumption. Transportation will depend entirely upon oil, and buildings will be 50 percent or more dependent upon coal. Projections of industrial energy consumption, however, appear to be less sensitive to price and conservation, and a switch to coal would probably at least double emissions. Again it is not clear how factors such as waste cycling of energy intensive material, increased product quality, tax structures and rebates, technological innovation, trends in consumer demand, and the development of a conservor ethic will affect industrial energy consumption and coal consumption. It is conceivable that industrial energy consumption could pose an increased emission threat when total energy demand decreases.

Suggested research topics and demonstrations fall into two groups: Modeling with emphasis on environmental impact, and technical studies. A total energy modeling effort is needed for various scenarios, focusing on environmental emissions rather than overall energy demand. An important area to examine in detail is the industrial sector. Specific activities should be broken down as to probable emissions, and scenarios developed based upon alternate energy sources, alternate product demand, and possible technological innovation, particularly in material substitution. The assumption that coal will replace oil and gas should be examined critically as this assumption appears to be a major determinant of projected emissions. In addition, special incentives such as taxation (on coal) and tax rebate (for pollution emission) should be considered as to their bearing on emission reductions. Finally, a scenario focusing on environmental emissions needs to be developed for a "conservor" society with changed lifestyles; presumably the results of this study would project minimum emissions in the future.

Other research programs involve research and demonstration studies. It is suggested that demonstration studies be considered as "contests" in which individuals, industry, and communities could compete. The rewards could be tax rebates or direct funding.

Some areas of effort include (1) dwelling and building design emphasizing space factors; (2) district heating development; (3) designing and using various tax rebates to promote energy efficiency for individuals, industry, and communites (Hirst, 1979); (4) community and industrial design development to minimize transportation requirements; and (5) replacement of energy intensive and cold-dependent processes, products, and uses in home and industry with alternates.

There appears to be a need for more research emphasis on coal and coal wastes with the focus on environmental abatement. The feasibility of using fly ash and other wastes for sulfur gas scrubbing needs to be studied.

These are but a few suggestions for research and development projects. The main thrust here is to focus on atmospheric emission reduction rather than energy conservation as a goal. Pricing, alternate approaches, and utilization of waste (Spilhous, 1970) need to be emphasized in context to existing studies. Most of the implementation of the above proposals is long term; however, the market place can respond amazingly fast, given the right incentives.

ATMOSPHERIC POLLUTANT TRANSPORT

Long range transport is the particular phenomenon associated with acid aerosols and acid precipitation. Particular reference has been given to tall stacks in this regard. It is important to ascertain the sensitivity of stack height in a particular setting to the development of acid aerosols and transport. By defining the key sources, this sensitivity could be used to develop a priority for abatement by scrubbing.

Modeling appears to be the best approach to developing a stack height sensitivity and acid aerosol formation and transport (Fischer, 1979). Lofting factors which include vertical stratification, thermal rise diffusion, and transport relative to the kinetics of SO_2 oxidation need to be emphasized. The master variable given a specific setting would be stack height.

This research is needed now. There is sufficient technical information and models available to carry out the effort. Calibration of the model using aircraft should be carried out.

CHEMICAL TREATMENT

Chemical modification of acid lakes from treatment is a short-term abatement effort. It is generally suitable for research purposes, but it may be feasible for certain other lakes also. Studies have focused on a specific lake except in Norway where parts of rivers have also been treated. Hydrologic modifications using ground water may be feasible in certain areas where a more permanent acidification abatement may result.

Basic materials ($Ca(OH)_2$, $CaCO_3$, $CaMg(CO_3)_2$,Ca, Mg-silicates have been added to lakes and the surrounding land to increase the pH and the acid neutralizing capacity. Various neutralizing materials have been studied (Grahn and Hultberg, 1975); these materials have generally been $Ca(OH)_2$ or $CaCO_3$ and they are usually added directly to the lake (Scheider and Dillon, 1976; Wiklander, et al, 1972). These studies have used one lake or a small portion of a drainage system.

The chemical treatment of acid lakes can be considered as a titration of water and surrounding sediment in contact with the water. Treatment of water only is temporal due to the runoff of the water containing the acid neutralizing capacity. A more efficient and permanent treatment would be to treat the surrounding soil to increase the base saturation. In this case the acid neutralizing capacity stays in the soil to react with the acid precipitation.

Many drainage basins with soils of low exchange capacity have the upper 10 m or so depleted in base saturation, but deeper layers may contain base saturation or acid neutralizing capacity (above pH 5.5 critical to lakes). This phenomenon results from the depletion of base saturation by the acid weathering of soils over the past 10,000 years or so. In this situation, there may be a large acid neutralizing capacity of soils at depth, but there is little contact of surface waters with these deeper soils. Therefore, hydrologic engineering can cause permeation of deeper soil zones and the neutralization of acid precipitation. This kind of treatment would consider an entire drainage basin in most cases. It is worthwhile noting that little information is known about the chemical properties of soils below the surface layer.

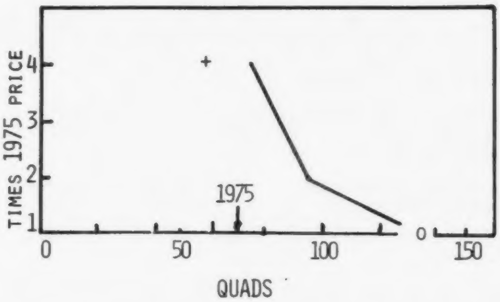

Figure 1. — Total energy demand from CONAES scenarios for 2010. + — A* scenario (4 times price plus conservation). O — B' scenario, 3 percent GNP growth. Compare to 1975 energy demand.

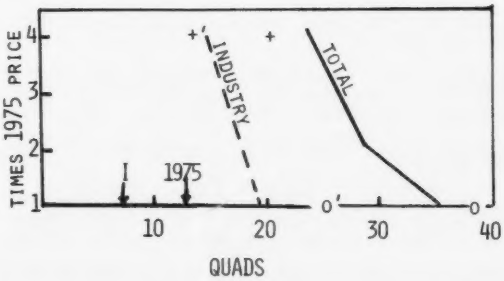

Figure 2. — Coal energy demand. Symbols same as in figure 1, except primes refer to industry. Note increase in A* estimate (+) over present demand.

SUMMARY

Energy conservation may not be directly coupled to emission decreases. Research using energy models focusing on emissions and sources of emissions is needed. Variations in the amount and kind of energy used and the resulting emissions can probably equal the 90 percent reduction attainable by scrubbing of emissions. Contests are suggested to implement innovation and change.

Research on emission scrubbing should focus on the use of solid wastes from coal as well as the exploitation of metals in these wastes.

Research on short-term abatement can identify important emission stacks using modeling to predict sensitive stack height.

In chemically treating acid waters, aquatic water systems should focus on the entire drainage basin and especially soils. Deep soils and sediments have sufficient neutralizing capacity in many cases to neutralize acid precipitation.

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MUTUAL RELATIONSHIP pH/EUTROPHICATION — ACID RAIN

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ABSTRACT

In older literature the pH of lake waters was used in attempts to quantify the eutrophication process. Because most of the data were taken from Swiss lakes the results were really only valid for hard waters. In addition, the influence of the temperature on the differences between summer and winter values of pH was not sufficiently taken into account. In eutrophic lakes, two quite different processes may take place after an increase of the pH value. In hard waters in which calcium concentration may control the phosphate solubility the formation of apatite (calcium phosphate) will counteract the eutrophication by withdrawal of phosphate from solution. If, on the other hand, the phosphate concentration is controlled by ferric hydroxide - as suspended clay component or as free hydrated ferric hydroxide - an increase of the pH may solubilize phosphate from the sediments, stimulating eutrophication. Two interesting processes make a theoretical approach of the calcium carbonate system extremely difficult: (a) the occurrence of $CaCO_3$ supersaturation has been known for a long time. Recently, however, it was found, that the degree of supersaturation is related to the pH. (b) Diffusion of CO_2 into the lake seems to be a much more complicated process than simple models predicted, as wind stress and microstratification in the lake depend more on physical-climatic factors than can be theoretically quantified. The combination of the pH-eutrophication relationship with the acid rain problem causes interesting thoughts for speculation. In poorly buffered, soft waters, the increase in acidity will decrease the availability of carbon. The waters may easily become carbon limited, especially if heavily fertilized, unjustifiably reviving the old carbon-phosphate controversy. In hard waters, the acid rain may decrease the pH value relatively little: but such a pH decrease may render the apatite more soluble and thus more available. Increased acidity in rain may in certain rock areas or types increase phosphate erosion.

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AN EVALUATION OF METHODS FOR MEASURING THE GROUNDWATER CONTRIBUTION TO PERCH LAKE

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ABSTRACT

Efforts to determine elemental fluxes within lakes have often been limited by incomplete knowledge of inputs and outputs of water and solutes. The major problem in determining lake budgets is the ground water component. In this paper we summarize the results of five methods used to estimate the volumetric flow of ground water to a small (45 hectare) lake in Eastern Ontario. The methods are : (1) a water-budget method, which uses estimates of evaporation, change in storage, and surface inflows and outflows to calculate the *net* groundwater inflow; (2) a Darcy approach involving estimates of hydraulic gradient, hydraulic conductivity, and area of aquifers communicating with the lake; (3) a chemical method, which is based on the isotopic dissimilarity between ground and surface water; (4) a point-dilution method, which is similar to the Darcy approach but uses groundwater velocity rather than hydraulic conductivity and gradient; and (5) a seepage-meter method, in which fluxes across the sediment-water interface are measured directly. None of the methods is completely satisfactory, and in most cases, investigators may have to use more than one approach. None of these methods adequately solves the problem of nutrient exchanges between ground water and lakes because of the reactivity of most nutrients with the sediments through which seepage occurs.

INTRODUCTION

The "invisible" contribution of ground water is the most difficult to measure, and sometimes the most mysterious, of all the components of a lake water budget. When attention is given to surface water quality, water resource managers often require information on the nature and volume of all inflows and outflows. Few managers can shake the gnawing feeling that the groundwater component is inadequately understood. Indeed, some workers (Uttormark, 1974; Lee et al. 1980) have suggested that ground water may flush nutrients from sediments into the overlying water. And Cartwright, et al. (1979) have found upward groundwater flow potentials in the littoral and pelagic sediments of Lake Michigan. Where contaminants are present in ground water (as Love Canal publicity reminds us) concerns are directed toward finding and monitoring groundwater discharge locations.

Our purpose here is to present an evaluation of work done on the groundwater component of a small lake (0.45 km^2) in eastern Ontario. This evaluation is based largely on work presented in detail elsewhere (Barry, 1975).

The methods compared in this report are:

1. Water budget from surface hydrology measurements.
2. Classical hydrogeologic method based on the Darcy equation.
3. Stable isotope ratios.
4. Point dilution.
5. Seepage-meters/mini-piezometers.

Table 1 summarizes the equipment needed and the variables to be measured for each of these methods.

Surface hydrology — The water balance equation may be written:

$$\Delta S = I - O + P - E \pm G \qquad \text{(eq. 1)}$$

where ΔS is the change in amount of water stored in the lake

I the surface-water inflow
O the surface-water outflow
P the precipitation
E the evaporation and
G the *net* groundwater flow.

Because all terms except E and G are directly measurable, independent estimates of either E or G make it possible to estimate the other. For Perch Lake, with values of E available from detailed energy-budget measurements (Barry, et al. 1979), the water budget was used to calculate G. Unfortunately, the available values of E were obtained when the lake was free of ice (i.e., May-October). An evaporation value (10.6 cm-yr^{-1}) for the period of ice cover was obtained from Bruce and Weisman (1967) and from the assumption (also from Bruce and Weisman, 1967) that the long-term average annual evaporation equals 1.18 times the open-season evaporation.

Average water budget figures for Perch Lake (Table 2) indicate that the annual groundwater inflow (2.89 x 10^{5}m^3) is of the same order of magnitude as the direct precipitation (3.63 x 10^{5}m^3) and the evaporation (3.14 x 10^{5}m^3). Ground water contributes 14 percent of the total annual inflow.

Table 1. — Equipment and variable requirements for the five methods.

Streamflow weirs, net radiometers, hygrothermographs, rain and snow gages, water temperature probes, water-level	water budget	continuous records of streamflow, precipitation, evaporation, lake level, lake morphometry, vertical profiles of water temperature, relative humidity, air temperature, net radiation
Observation wells and piezometers, drilling rig, water pumps, water-level recorders	classical hydrogeologic	weekly groundwater levels, estimates of hydraulic conductivity and aquifer thickness around the lake
Streamflow weirs, evaporation pan, rain gage, piezometers, mass spectrometer	stable isotope	$^{18}O/^{16}O$ ratio streamflow, ground water precipitation, evaporation pan, and lake
Drilling rig, borehole wells, tracers, tracer-dectector, downhole mixing pump	point dilution	distribution of the volumetric flux of ground water, aquifer thickness around the lake
Seepage-meter*, drive casing hammer, drive points, plastic tubing, plastic bags	seepage-meter/mini-piezometer	groundwater discharge distribution across lakebed, hydraulic conductivity

*Described in text

The net groundwater inflows on a monthly basis are shown in Figure 1. With spring rains and snowmelt, groundwater flow increases rapidly and reaches a peak in April. During May, evaporation increases as the vegetation leafs out and groundwater flow declines through the summer. In September with the onset of autumn rains, the first killing frost, and declining vegetation, groundwater rates begin to increase. They reach a maximum in November when recharge ceases as rain gives way to snow. Surface flows display similar seasonal changes, probably for the same reasons. However, the contribution of ground water to the total inflow to the lake varies from a low of less than 10 percent in May to a high of 30 percent in August and September.

Surface hydrology provides the most obvious and most readily accepted method of estimating the groundwater component. However, this method suffers from several problems:

1. Only the net groundwater flow is determined. It tells nothing about the source areas or significant leakages through the lakebed.
2. The groundwater flow is a residual term which, for many lakes, is represented as a small difference between large numbers.
3. Energy budgets, required for the evaporation term, and surface flow monitoring are expensive.

Hydrogeologic or Darcy approach — Darcy's equation may be written:

$$Q = KA\frac{\Delta h}{\Delta x}$$

where Q is the volumetric groundwater flow, A, the cross-sectional area of the flow path, h/ x, the hydraulic gradient along the flow path, and K, hydraulic conductivity of the geologic material.

Determining the pattern and rates of groundwater flow into a lake requires knowledge of (1) the vertical cross-sectional area of aquifer materials that transmit water to the lake, and (2) the distribution of hydraulic head in that vertical cross section. Hydraulic head is measured with piezometers and observation wells. In the example shown in Figure 2, the piezometer water levels are higher with greater depth. This indicates there is an upward component in the groundwater velocity at the site. Decreasing water levels with depth would indicate downward flow.

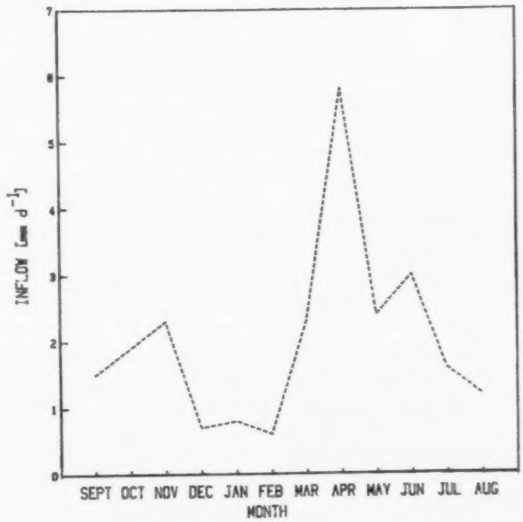

Figure 1. — Monthly groundwater flow to Perch Lake from surface water budgets 1970-1977.

In the Perch Lake study it was possible to employ the classical hydrogeologic or Darcy method on the subbasin aquifer contributing a substantial share of ground water to the lake. Hydraulic conductivities obtained by several standard methods at over 65 locations in a land area of 0.5 km^2, were judged to have upper and lower estimates of 1×10^{-3} to 5×10^{-3} cm·s^{-1} for the sands and 1×10^{-6} to 1×10^{-4} cm·s^{-1} for the basal silt and clay.

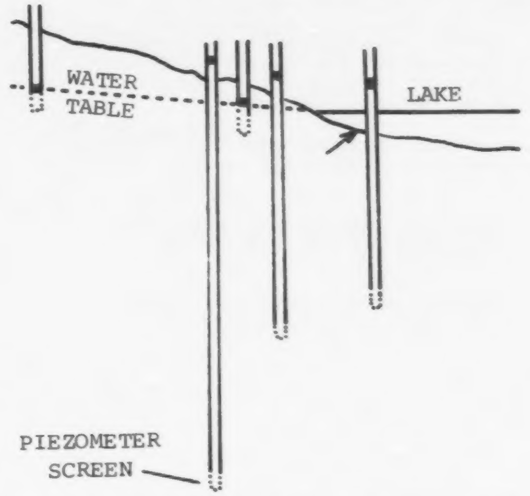

Figure 2. — Vertical section in the line of groundwater flow at the lakeshore. Dark horizontal bars in piezometers show water levels (hydraulic head) at the piezometer screens. A situation of increasing head with depth indicates groundwater flow potential into the lake.

Average annual hydraulic gradients were determined for various parts of the section. The representative gradients ranged from 0 to 0.034. The total calculated flow had an upper value of 364 $m^3 \cdot d^{-1}$ (or 0.91 $mm \cdot d^{-1}$ over lake surface) and a lower value of 73 $m^3 \cdot d^{-1}$ (0.18 $mm \cdot d^{-1}$). Even with the high sampling density used, data were insufficient to assign different gradients to different seasons or months.

The major difficulty with the classical hydrogeologic approach (Darcy method) is determining the magnitude and spatial variation of hydraulic conductivity. Although the work of Cherry, et al. (1975) at Perch Lake was one of the most thorough groundwater/lake investigations in an area of its size, it resulted in estimates of groundwater discharge that varied by a factor of 5.

Isotope method — Where the ground water is chemically different from the surface waters into which it flows, groundwater flow can often be estimated. The technique used at Perch Lake is based on the idea that the heavy isotope of oxygen, O, which is naturally present in water as $H_2{}^{18}O$ can vary relative to the H O because of different rates of evaporation and condensation. These processes cause isotopic fractionation that results in "fingerprinting" different water masses according to their $^{18}O/^{16}O$ ratios. At Perch Lake the ground water is isotopically lighter (higher $H_2{}^{16}O$ to $H_2{}^{18}O$, ratio) than the lake water. Different sources of ground water also differ in their O enrichment. However the major uncertainty in the Perch Lake study (Welhan and Fritz, 1977) l y in the $^{18}O/^{16}O$ ratio of water evaporating from the lake.

If the groundwater flow is small (as in Perch Lake), the isotope method probably has an error of up to 50 percent. Other obvious problems with the isotope method are:

1. The necessity for having a uniform isotopic ratio in ground water or a way to assign proportional amounts of groundwater inflow from isotopically different zones.

2. The necessity for a fairly large number of sampling points and information on geologic units that transmit water to the lake.

3. A sufficiently large isotopic difference between the surface and ground waters.

4. Of all the parameters that must be measured the isotopic composition of evaporating moisture is the most difficult and can result in an uncertainty of more than 50 percent in the calculated lake evaporation rate (Zimmerman and Ehhalt, 1970).

Point methods - Two techniques, which complement both the surface hydrology and the classical hydrogeologic approaches, are point-dilution measurements of groundwater velocity and seepage-meter/mini-piezometer methods. They complement these methods because they directly measure groundwater flux at specific points. These techniques are of particular interest where there are known or suspected sources of onshore groundwater contamination or where a lakebed-aquifer system is fairly homogeneous. Neither of these methods has been employed over an area wide enough in Perch Lake that groundwater inflow calculations can be given.

Seepage-meter and mini-piezometer methods - This approach relies on the fact that significant groundwater inflows tend to occur through sediment (peats, sands, gravels) in shallow nearshore areas. A seepage meter is a cylindrical enclosure on the lakebed to which a deflated submerged plastic bag is attached (Lee, 1977). Where groundwater inflow is upward, the flow is determined by measuring an increase in the water volume of the bag over a period of time, generally several hours. In fairly homogeneous systems the seepage rate declines exponentially with distance offshore (Lee, 1977). If a smooth pattern of seepage flux is found, measurement points can be used to estimate groundwater inflow through an area of lakebed (Lee, et al. 1980). Mini-piezometers and small bundle-type samplers are an inexpensive, manual method, useful for identifying zones of significant groundwater flow potential. They are also useful for sampling pore waters in cohesionless sediments of seepage zones. These samplers are installed simply by driving a ½ inch (nominal) steel pipe to the desired depth (4 m maximum), inserting the plastic sampling tube(s), and withdrawing the pipe (Lee, et al. 1980). As shown in Figure 2, zones of upward flow can be identified once equilibrium water levels are reached.

Point dilution measurements of groundwater flow - A critical review of this method was given by Halevy, et al. (1966). The technique consists of labeling the water in a well screen with a tracer and observing its rate of dilution. If the tracer solution is well mixed, the slope of the dilution curve (log concentration vs time) gives the rate of apparent groundwater flow.

The speed of the groundwater can be related to the rate of dilution through the equation:

$$v_t = \frac{V}{\alpha F t} \ln C/C_0$$

where v is the volumetric flux of the water through the screen

v the dilution volume

F the cross section of the well screen

t the time from the beginning of measurement

C the original concentration

C the observed concentration at t and

a correction factor for distortion of flow by the well screen.

The value of the point dilution technique can be illustrated by noting that apparently homogeneous sands can conduct groundwater flow at rates that vary by a factor of 5 (Pickens, et al. 1977). In most cases the methods for measuring hydraulic conductivity are not as sensitive as the point dilution technique.

Table 2. — Perch Lake water budget for 1970-77.

Source	Volume ($\times 10^5$ $m^3 \cdot yr^{-1}$)
Surface streams	
# 1	2.23
# 2	9.70
# 3	1.51
# 4	0.84
# 5	0.29
Surface stream inflow	14.57
Precipitation	3.63 (≡ 0.807 cm)
TOTAL IN	18.20
Surface stream outflow	17.95
Evaporation	3.14 (≡ 0.698 cm)
TOTAL OUT	21.09
Net ground water	2.89
(TOTAL IN — TOTAL OUT)	

Note: Evaporation measurements are from energy-budget calculations (Barry, et al. 1979).

Clearly this method and the seepage-meter method obviate the necessity of separately determining the hydraulic conductivity and the gradient. The point dilution method cannot distinguish readily the vertical and horizontal components of flow, nor does it indicate flow direction.

The fundamental limitation of point methods is the need to interpolate between bore holes or measurement locations on the lakebed since there are practical limitations on density of sampling points. In many lake settings, successful application of point methods in estimating groundwater inflow or outflow will probably require methods for characterizing aquifers by rapid remote sensing techniques.

COMPARISON OF METHODS

Table 3 provides a basis for comparing groundwater inflow estimates for the same 4-month period. The point dilution and Darcy estimates are constant because the calculations were based simply on average annual or representative values. These two methods were used on the northern side of Perch Lake, not the whole lake perimeter, so the estimate is expected to be low. Both the water budget and the stable isotope methods appear to agree. However, the uncertainties in the stable isotope method are large (±50 percent) relative to those of the water budget method (±10 percent) so the agreement may be coincidental. Because the northern side of the lake is soft peat, it would have been difficult to employ seepage meters there.

Table 3. — Comparison of estimates of groundwater inflow to the lake for the period May through September, 1973.

	Groundwater inflow to the lake in mm·da^{-1}*				
Method	May	June	July	August	September
Water budget	2.0	5.5	1.6	1.2	-
Stable isotope	-	6.4	0.35	1.2	0.43
Point dilution	1.0	1.0	1.0	1.0	1.0
Darcy°	0.9	0.9	0.9	0.9	0.9

*Values are daily volumetric water flow into the lake divided by lake surface area

°The groundwater inflow from the Darcy method is 0.2 or 0.9 for the lower and upper estimates of hydraulic conductivity.

CONCLUSION

There are essentially two types of lake/groundwater flux methods: Gross measurements which, by their nature, are averaged over large areas (the stable isotope and surface hydrology methods), and syntheses made from many point measurements (the seepage-meter/mini-piezometer, the classical hydrologic, and the point dilution methods). The major limitation of point measurements is geologic complexity which contributes to a wide spatial variation in flow rate. A most promising technique to allow interpolations between sampling points is the use of ground-probing radar that "sees" into the subsurface, particularly in coarse-grained soils (Annan and Davis, 1976).

The choice of methods will depend on:

1. The type of information needed;
2. The features of the study site, e.g., the presence of access roads or stream gaging structures; and
3. The manpower, skills, and equipment available.

All methods for groundwater measurement are expensive but it is unrealistic to appraise any costs until the study area and requirements are known. All methods require long times to get a stable mean flux. Precipitation, for example, varies widely from year to year. Changes in storage of energy and water in lakes are subject to considerable error on the short term (1 or 2 days).

One of the lessons at Perch Lake has been that it was necessary to compare methods if we were to avoid being deluded into thinking that one method gave a correct measurement. It was often necessary to go through an iterative process of checking one method against the other. The significance of ground water to lake processes should be studied by as many methods as feasible. With an environmental variable as complex as ground water, it could be quite misleading to put complete faith in any single method. Internal consistency among various methods has provided a basis for confidence. But none of the methods addresses the problem of nutrient fluxes due to ground water/surface water interaction.

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REHABILITATION PROJECT FOR A QUEBEC LAKE: WATERLOO LAKE, NEAR MONTREAL

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ABSTRACT

Lake Waterloo, a small shallow eutrophic lake (A, 1.5 km^2; $\bar{\bar{Z}}$ 2.9), is situated in the southeastern region of the Province of Quebec. This lake, which possesses a very small watershed (31.5 km^2), has been the object of a restoration program by the Ministry of the Environment since 1976. This program is divided into two distinct operations. The first one consists of installing an aeration system of the diffuser type; at the present time, it has been in continuous operation for 4 years. During this period no winterkill episodes have been observed because of the concomitant increase of mean dissolved oxygen values under ice cover (bottom: 8.0 mg l^{-1}). Parameters mostly affected by aeration were total iron (mean decrease 95 percent), total manganese (mean decrease 55 percent), ammonia (mean decrease 55 percent), and total phosphorus (mean decrease 42 percent). A study of the phytoplankton populations has shown a marked transition since 1978, from the previous dominant Cyanophycea towards Bacillariophycae. The second part of the overall project is a study of the total phosphorus input to the lake (1963 kg P yr^{-1}). Industrial and urban activities account for 50 percent of this and should be eliminated by 1982. Total restoration of this water body needs a supplementary method to diminish residual phosphorus originating from the watershed (884 kg P yr^{-1}). In this particular case, dredging the lake sediment (thickness 5.8 m) is the only way to achieve this objective even if the investment seems infeasible on a short-term basis.

INTRODUCTION

The installation of an aeration system in 1976 at Lake Waterloo was ecological intervention urgently needed to eliminate further winterkills. The continuation of this intervention is justified because artificial aeration of lakes can improve water quality and extend the vertical distribution of the biota. Numerous studies have shown an increase of dissolved oxygen concentrations (Irwin, et al. 1966; Haynes 1971). Noticeable decreases in the concentration of manganese and iron (Wirth and Dunst 1967; Haynes 1971), ammonia (Symons, et al. 1967) and hydrogen sulfide (Irwin, et al. 1966; Leach and Harlin 1970) have been observed in the deepest portion of such lakes. Changes in the biological populations have been observed in artificially mixed lakes, including a decrease of the phytoplankton populations (blue-green algal biomass, Anon. 1971; Malueg, et al. 1971), an extension of the vertical distribution of zooplankton (Fast 1971), and an increase in number and speciation of the benthic macroinvertebrates. The second phase of the program is synthetic and corresponds to the estimation of the allochtonous and autochtonous phosphorus budget. This should allow us to determine the restoration techniques having the highest probability of success.

METHODS

Aeration

Figure 1 shows the locations of the diffusers, the sampling station used to interpret the physico-chemical data, as well as certain morphometric parameters. The samples were collected monthly during winter and bi-monthly for the summer, at the surface (0.5 m) and at the bottom (3.5m), and analyzed using standard government laboratory methods. The results of the physico-chemical parameters were analyzed for every year of the aerator's operation. The efficiency of the aeration device was determined by comparing the means for 1975 (before aeration) with those for the period 1976 to 1979 inclusively. The statistical significance of the differences between the means was determined using the student t test.

Phosphorus budget

The phosphorus budget was estimated by three different methods. The first was an indirect estimation of the phosphorus inputs using available data from existing land use maps. The choice of the phosphorus exportation coefficients is in agreement with the literature (e.g; urban zones — 105 kg P km^{-2} yr^{-1}, Potvin 1976; swamps — 25 kg P km^{-2} yr^{-1}, Uttomark, et al. 1978) and permits a preliminary quantification of this watershed nutrient output. The second is more direct because it evaluates *in situ* the total phosphorus load from the tributaries discharging into the lake as well as anthropogenic point emissions in the vicinity of the lake. The data were collected bimonthly between September 1975 and September 1976 and permitted a more precise evaluation of the phosphorus load. Lastly, the autochtonous phosphorus input from the oxygen deficient ($\leq$ mg l^{-1}) sediments was established using a releasing coefficient of 8.0 mg P m^{-2} day^{-1} (Kamp-Nielsen, 1974; Fekete, et al. 1976).

RESULTS AND DISCUSSION

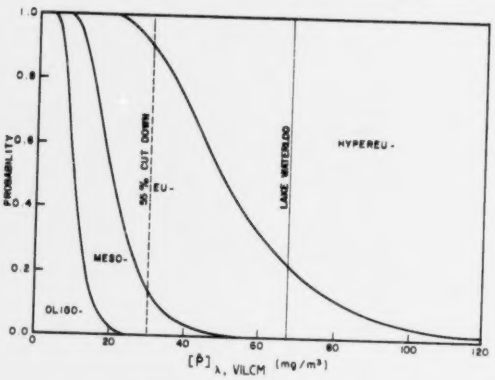
Aeration

Diffuser type aeration systems normally produce convection currents which destratify water bodies. This mixing of the water column may provoke an increase in temperature. From Table 1 we can see that such a phenomenon has not occurred; on the contrary a significant cooling is observed (Table 2). The dissolved oxygen concentration has increased by 20 percent near the bottom and no oxygen deficit has been detected since the winter of 1977, eliminating winterkill episodes. The oxygen saturation levels are similar to the dissolved oxygen values (Tables 1 and 2). The transparency of the water column did not change as expected, because of the mixing by the diffusers. It would seem that this parameter is influenced by biological populations such as phytoplankton and zooplankton. The lowering of the pH is significant for 1976-1979 period and has occurred in the entire water column (Tables 1 and 2).

These results were predictable if we consider the fact that fermentation processes have been replaced by heterotrophic oxidation, producing CO_2. The increase in the concentration of the CO_2 is attributable to the nitrification of ammonia and the oxidation of sulfates. This phenomenon reduces pH values which is less perceptible within the upper water layer because of its bioassimilation by the phytoplankton. The soluble iron concentration increased for the aeration period (Table 1) but these differences are statistically non-significant (Table 2). The change of the soluble ferrous ions previously released from anoxic sediments into an insoluble ferric hydroxide ($Fe(OH_3)$) has resulted in a marked decrease (95 percent) of the total iron concentration of the water-sediment interface. Low manganese values be they soluble or total are significant (Table 2).

As in the case of iron and sulfates an increase in the redox potential (En) has caused the precipitation of compounds such as manganese carbonate ($MnCO_3$), manganese sulfide (Mn S), and manganese hydroxides (Mn (OH_2)). Magnesium did not show major fluctuations which is understandable since it rarely precipitates out. The organic phosphorus concentration remained constant during the aeration period. This is interesting since the transition from fermentation to oxydation of the organic matter did not increase the concentration of this labile substance.

Table 1 indicates that before aeration an active unidirectional flux of inorganic phosphorus resulted in a continuous enrichment of this water body. The inorganic phosphorus flux was stopped and probably reversed with the regeneration of an oxidized micro-zone at the water-sediment interface. The formation of



$$[\hat{P}]_{\lambda}\ VILCM = \frac{[P]_{\circ}}{1+\sqrt{\tau_w}}$$

$[\hat{P}]_{\lambda,\ VILCM}$ = PREDICTED CONCENTRATION OF TOTAL PHOSPHORUS

$[P]_{\circ}$ = PHOSPHORUS INFLOW CONCENTRATION

τ_w = WATER RESIDENCE TIME

Figure 2. — Probability of a prediction falling within a particular trophic class.

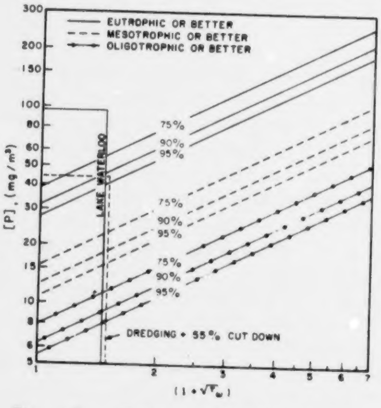
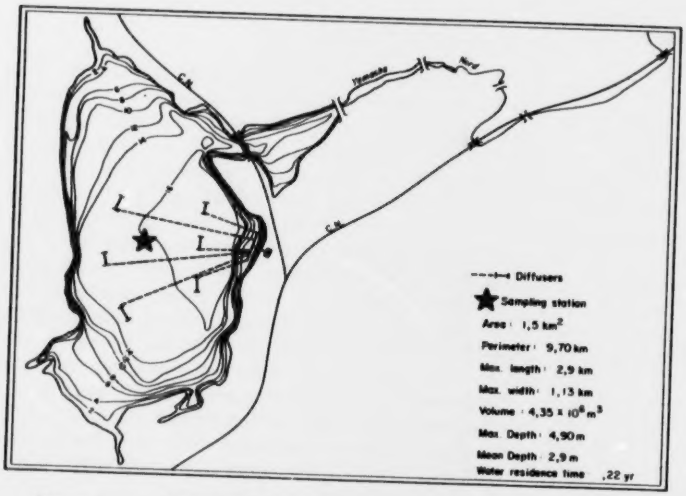


Figure 3. — Probabilistic loading plot showing the logarithm of the predicted inflow concentration as a function of the water residence time. Percentages represent the certainty of the effectiveness of the inflow concentration achieving the expected trophic state.



Area : 1,5 km²
Perimeter : 9,70 km
Max. length : 2,9 km
Max. width : 1,13 km
Volume : 4,35 x 10⁶ m³
Max. Depth : 4,90 m
Mean Depth : 2,9 m
Water residence time ,22 yr

Figure 1. — Lake Waterloo: Sampling station and morphological data.

co-precipitates of phosphates with iron, manganese, and carbonates resulted in a significant (Table 2) reduction in the concentration of inorganic phosphorus for the entire water column. The means (Table 1) for total phosphorus reflect the pathways controlling the inorganic and organic phosphorus concentrations.

When anoxic zones were detectable (before 1975), bacterial nitrification by which ammonia is progressively oxidized into nitrites and nitrates was inhibited. The concomitant decrease in redox potential resulted in the accumulation of NH_4^+ - NH_3 concentration by the bacterial population; this did not cause an increase in NO_2^- - NO_3^-. The reduction in the concentration of organic nitrogen may be imputed to the increase in the absorption capacity of the sediments as a result of the restoration of an oxidized microzone. The increase in

Table 1. — Mean and Standard deviation of the different parameters sampled at lake Waterloo from 1975 to 1979.

Parameters	Surface*						Bottom**					
	1975			1976-1979			1975			1976-1979		
	$\bar{X}$	σ_x	n***	$\bar{X}$	σ_x	n***	$\bar{X}$	σ_x	n***	$\bar{X}$	σ_x	n***
Temperature °C	15.4	7.76	8	11.7	8.55	38	14.1	6.92	8	11.6	7.67	38
Dissolved Oxygen mg $1^{-1}O_2$	10.1	1.16	8	10.0	1.94	38	6.6	3.29	8	8.0	2.66	38
Oxygen Saturation %	98.54	12.72	8	92.13	23.49	37	57.11	32.85	8	74.35	25.48	37
Transparancy m.	1.02	0.29	7	1.1	0.49	29	--	--	-	--	--	--
pH	8.3	0.82	8	7.5	0.96	38	7.6	0.72	8	7.2	0.72	38
Soluble Iron mg 1^{-1}Fe	0.06	0.02	7	0.07	0.04	31	0.10	0.05	7	0.10	0.08	31
Total Iron mg 1 Fe^{-1}	0.15	0.05	9	0.17	0.07	37	0.40	0.27	9	0.02	0.06	37
Soluble Manganese mg 1^{-1}Mn	0.06	0.01	7	0.04	0.05	31	0.10	0.05	7	0.06	0.06	31
Total Manganese mg 1^{-1}Mn	0.07	0.01	9	0.07	0.03	37	0.20	0.17	9	0.09	0.06	37
Magnesium mg 1^{-1}Mg	2.2	0.07	9	2.2	0.34	31	2.3	0.15	9	2.2	0.46	31
Organic Phos.mg 1^{-1}P	0.017	0.012	9	0.015	0.011	37	0.018	0.009	9	0.014	0.013	37
Inorganic Phos.mg 1^{-1}P	0.018	0.011	9	0.012	0.004	37	0.027	0.022	9	0.011	0.004	37
Total Phos. mg 1^{-1}P	0.036	0.013	9	0.027	0.013	37	0.045	0.027	9	0.026	0.013	37
Organic Nitrogen mg 1^{-1}N	0.73	0.38	8	0.47	0.29	37	0.63	0.37	8	0.47	0.26	37
$NO^2 + NO^3$ mg 1^{-1}N	0.18	0.37	9	0.12	0.15	37	0.17	0.34	9	0.11	0.14	37
NH^4 mg 1^{-1}N	0.04	0.04	9	0.06	0.06	37	0.15	0.18	9	0.08	0.08	37
Organic Carbon mg 1^{-1}C	14.1	4.0	9	15.5	8.88	34	14.5	3.83	9	14.7	7.60	34
Inorganic Carbon mg 1^{-1}C	3.9	1.13	9	5.8	2.33	34	4.7	0.83	9	6.1	2.41	34
Total Carbon mg 1^{-1}C	18.0	3.88	9	21.8	7.96	34	19.2	3.29	9	20.8	7.41	34
Phytoplankton Biomass mg m^{-3}	3634.50	1852.38	8	7189.27	10741.62	31	--	--	-	--	--	--
Total Chlorophyll — a mg m^{-3}	41.28	31.50	7	31.98	27.32	34	--	--	-	--	--	--

* Surface = 0.5 m
** Bottom = 3.5 m
*** (n) = number of samples

Table 2. — Comparison of the means of the different parameters using student (t) test between 1975 and 1976-1979 at lake Waterloo.

Parameters	Student (t) Test 1975 vs 1976-1979			
	Surface*		Bottom**	
	Significantly Different (95%)***	Not Significantly Different (95%)***	Significantly Different (95%)***	Not Significantly Different (95%)***
Temperature °C	X			X
Dissolved Oxygen mg $1^{-1}O_2$		X	X	
Oxygen Saturation %		X	X	
Transparency m.		X	-	-
pH	X		X	
Soluble Iron mg 1^{-1}Fe		X		X
Total Iron mg 1^{-1} Fe		X	X	
Soluble Manganese mg 1^{-1}Mn	X		X	
Total Manganese mg 1^{-1}Mn		X	X	
Magnesium mg 1^{-1}Mg		X		X
Organic Phos.mg 1^{-1}P		X		X
Inorganic Phos.mg 1^{-1}P	X		X	
Total Phos. mg 1^{-1}P	X		X	
Organic Nitrogen mg 1^{-1}N	X		X	
$NO^2 + NO^3$ mg 1^{-1}N	X		X	
NH^4 mg 1^{-1}N	X		X	
Organic Carbon mg 1^{-1}C		X		X
Inorganic Carbon mg 1^{-1}C	X		X	
Total Carbon mg 1^{-1}C	X			X
Phytoplankton Biomass mg m^{-3}		X	-	-
Total Chlorophyll — a mg m^{-3}		X	-	-

* Surface = 0.5 m
** Bottom = 3.5 m
*** (95%) = Confidence level

inorganic carbon concentration is closely linked to the increase in dissolved oxygen and has been discussed previously. As for organic carbon, the small increase of the surface concentration may be attributed to a plankton biomass increase. The phytoplankton biomass shows *a* non-significant (Table 2) increase for the aeration period which is not concomitant to the chlorophyll *a* values. Since the cellular concentration of chlorophyll *a* varies from species to species (Wetzel, 1975) it is normal to observe such results because a species shift in the phytoplankton population has been occurring since 1978 (Cyanophycae towards Bacillariophycae, Choquette, 1979).

Phosphorus budget

If we refer to Table 3 we notice that the direct and indirect methods for determining annual allochtonous phosphorus inputs are comparable and represent a load of 1,963 kg P yr^{-1}. Estimation of the annual autochtonous phosphorus inputs coming from the sediments corresponds to approximately 35 percent of the total load originating from the watershed. The aeration of Lake Waterloo has theoretically inhibited nutrient flux from the sediments. Figures 2 and 3 show the actual trophic state following the probabilistic expression of Chapra and Reckhow (1979). By 1982, 55 percent of the total input (1,079 kg P yr^{-1}) will be eliminated. This cutback of 55 percent will bring the mean phosphorus concentration into the lake to 45 mg m^{-3} which should not produce marked modifications in the visual aspect of the lake (fig. 2) The penultimate solution seems to be a lake deepening operation that could effectively buffer the residual phosphorus loading (884 kg P yr^{-1}). Dredging 3,000 m^3 by suction would cost approximately $3,000,000 and would augment the water volume by 23 percent, but this would not assure a definitive restoration of Lake Waterloo (Figure 3: $1 + \sqrt{t_w} = 1.5$). Other dredging techniques are presently being studied to lower the cost of sediment extraction (e.g.; bulldozer, etc.)

Table 3. — Phosphorus budget in Lake Waterloo.

ASSESSMENT METHODS*	Loading kg P yr^{-1}
Indirect Estimation	1963
Direct Evaluation	1991
Sediments	686

SOURCES	IMPORTANCE %
Industrial	21
Stockbreeding	18
Fertilizer	3
Domestic uses	37
Forests & Rain	21

* Provencher et al., 1979.

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QUANTIFICATION OF ALLOCHTHONOUS ORGANIC INPUT TO CHEROKEE RESERVOIR: IMPLICATIONS FOR HYPOLIMNETIC OXYGEN DEPLETIONS

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ABSTRACT

Cherokee Reservoir was created by the Tennessee Valley Authority in 1947 as a multipurpose reservoir to provide flood control, power generation, and recreation. It is the largest of five TVA impoundments in the Holston River Basin of upper East Tennessee. Water releases from Cherokee Reservoir have been documented to be low in dissolved oxygen content since 1950. From 1970 to 1978, water released during power generation had less than 5.0 mg/1 DO, an average of 149 days per year, and less than 1.0 mg/l an average of 45 days per year. Previous studies strongly suggest that inputs of allochthonous organic material originating in the highly productive aquatic macrophyte beds in the Holston River below Kingsport, Tenn. adversely impact hypolimnetic DO concentrations in Cherokee Reservoir. The present study indicates that the annual average net primary production of aquatic plants in the Holston River above Cherokee Reservoir is 16.6 metric tons/ha/year (dry weight), a rate much higher than reported for rivers in the temperate regions of North America. Biomass contribution from this reach of river is estimated at 4,570 metric tons dry weight to Cherokee Reservoir annually. Deposition, subsequent decomposition, and nutrient release from this large amount of allochthonous aquatic macrophyte input represents 94 metric tons nitrogen, 10 metric tons phosphorus, and 4,570 metric tons biochemical oxygen demand, all significant factors in hypolimnetic oxygen depletion in Cherokee Reservoir.

INTRODUCTION

Cherokee Reservoir was created in 1941 as a multipurpose reservoir to provide flood control, power generation, and recreation. It is the largest of five Tennessee Valley Authority impoundments in the Holston River Basin northeast of Knoxville, Tenn. (Figure 1). The reservoir has a surface area of 131 km^2, an average depth of 15 meters and a mean hydraulic retention time of 178 days. Thermal stratification begins in mid-April and the use of hypolimnetic waters for power generation results in the reservoir becoming isothermal (24°C) by late summer. During early to midsummer, the reservoir characteristically has a shallow, highly productive epiliminion and a thick, oxygen deficient hypolimnion. Consequently, water released for power generation has less than 5.0 mg/l dissolved oxygen an average of 149 days per year and less than 1.0 mg/l an average of 45 days per year (1970 to 1978).

The problem of low DO content in water released for power generation became evident in 1950 (9 years after closure), when the reservoir discharged water with a DO content < 1.0 mg/l for a period of 38 days. The historical trend in the number of days with DO < 1.0 mg/l in the Cherokee Dam discharge is shown in Figure 2. Higgins (1978) concluded that the DO levels from 1963 to 1976 corresponded with increasing industrial productivity and wastewater discharges between 1960 and 1968, followed by a period of reduced discharge after 1969 because of plant closures, reduced industrial productivity, and increased wastewater treatment.

Between 1950 and 1979, over 25 water quality studies were conducted on Cherokee Reservoir. An overview of these studies is given by Iwanski (1978). Several studies were conducted because of the low DO problem. Of the studies reviewed, only three (Churchill and Nicholas, 1966; Gordon, 1971; and Iwanski, et al. 1980) have investigated in any detail the mechanisms causing hypolimnetic DO depletion.

Churchill and Nicholas (1966) observed that the DO concentrations in the Cherokee Reservoir outflow began to approach zero earlier each spring from 1963 to 1966 and concluded that this was primarily caused by high concentrations of nutrients entering the reservoir as inflow from the Holston River. They also suggested that prolific growths of aquatic weeds in the Holston River between Cherokee Reservoir and Kingsport, Tenn. adversely impacted the water quality of Cherokee Reservoir. They stated, "This heavy organic load, flowing continuously into Cherokee undoubtedly has an important part to play in the depletion of DO in the hypolimnion of Cherokee Reservoir."

Gordon (1971) investigated several different mechanisms of oxygen depletion in Cherokee Reservoir. He concluded that nitrification caused over 50 percent of the oxygen loss in the hypolimnion for the 1967-1970 period. While these studies did not directly address the impact of allochthonous organic matter on Cherokee Reservoir, several observations support the premise that aquatic macrophytes are a significant factor in hypolimnetic DO depletion.

Using 1970 data and a computerized reservoir hydrodynamics model from the Massachusetts Institute of Technology, Gordon (1971) also determined that flow to Cherokee Reservoir enters as an interflow; about 80 percent of the time, from mid-April to late September. In addition to *in situ* DO depletion mechanisms, the well-oxygenated water initially trapped under the thermocline in the deeper end of the reservoir at the onset of stratification eventually is discharged through the power turbines and is replaced by poorly oxygenated water from upstream reaches and interflow, thus reservoir hydraulics are a major factor contributing to the low hypolimnetic DO values in the lower end of Cherokee Reservoir. Gordon also demonstrated that oxygen depletion first begins and is most rapid between river miles 70 and 95 at the upper end of the reservoir, and that ammonia increased in the hypolimnion (after DO depletion) as a result of anaerobic deamination of the highly organic sediments.

Gordon's conclusions concerning interflow validate the hypothesis of Churchill and Nicholas who noted "...it seems likely from observed river and reservoir temperature data that some of the intermittently colder masses of inflowing waters have entered the head of Cherokee pool as interflows, some possibly entering below the thermocline." The drift of aquatic plants into Cherokee Reservoir was related to this thermal discontinuity by Hall (1966) who observed that "Between Cherokee boat dock (HRM 93) and the powerline crossing downstream (HRM 90), the detached, floating plants were lined up more or less perpendicularly to the axis of the old river or, in other words, formed more or less a line across the reservoir. It is wondered if the transverse accumulation of floating plants represents the approximate location at which the Holston River 'dives' under Cherokee Reservoir." Gordon's data and these observations indicate that the earliest and most rapid DO depletion occurs in the same area where the thermal discontinuity between inflow waters and Cherokee Reservoir would allow the deposition of allochthonous organic material.

The study by Iwanski, et al. (1980) concluded that the major causes of DO depletion and eutrophication in Cherokee Reservoir are inflows of phosphorus, nitrogen, BOD$_5$, and volatile suspended solids. Using the Water Quality River-Reservoir Systems (WQRRS) model (U.S. Corps of Engineers, 1977) and 1978 data, Iwanski, et al. conducted a sensitivity analysis and simulated the effect of these key factors on DO depletion in Cherokee Reservoir. This analysis showed that 37.5 percent of the annual DO depletion could result from inflow detritus (volatile suspended solids) alone and when combined with the temperature effect could account for over 60 percent of the annual DO depletion. Dissolved organic carbon (BOD$_5$), total dissolved nitrogen, and total dissolved phosphorus accounted for 15.0, 6.4, and 5.0 percent respectively, of the annual DO depletion, according to this model.

These studies strongly suggest that inputs of allochthonous organic material originating in the highly productive aquatic macrophyte beds in the Holston River adversely impact hypolimnetic DO concentrations in Cherokee Reservoir. However, lack of adequate data on aquatic macrophyte productivity and drift characteristics in the Holston River have precluded accurate assessment of the impact of aquatic macrophytes on DO regimes in Cherokee Reservoir. This paper reports the results of work in progress designed to obtain this data.

METHODS AND MATERIALS

Study Area

The Holston River is formed by the confluence of the North and South Fork Holston Rivers at Kingsport, Tenn. Flow rate in the Holston River is partially regulated by Fort Patrick Henry Dam, located on the South Fork Holston River approximately 5 miles above its confluence with the North Fork. In accordance with an agreement with the Tennessee Eastman Company, TVA releases a minimum daily average flow of 21.2m^3/sec from the dam to maintain an adequate water supply for the company. Bihourly flow records (1978) for the U.S. Geological Survey station at HRM 118.4 show the total flow varies from a minimum of 23.8 m^3/sec in October and December to a maximum of 1,209.1 m^3/sec in March. Instantaneous hourly flows during late summer and fall can vary from zero to 170 m^3/sec depending on power generation schedules and the flow from the unregulated North Fork Holston River. According to Iwanski, et al. (1980), total waste loads to the Holston just above Cherokee Reservoir (HRM 103.4) were measured to be 14,746 kg/day total nitrogen, 1,145 kg/day total phosphorus, and 10,024 kg/day BOD$_5$. Point source discharges accounted for 18 percent of the total nitrogen load, 65 percent of the total phosphorus load. Land use in the watershed of Cherokee Reservoir consists of 55 percent forest, 5 percent urban, 37 percent agriculture, and 3 percent other uses.

The primary study area was from HRM 141.2 (confluence of North and South Fork Holston Rivers) to HRM 109.1 (backwaters of Cherokee Reservoir). In this reach the Holston River has a gradient of 0.6 meters/km, surface area of 497.2 hectares, water depth of 0.3 to 3.5 meters, and a width of 106 to 203 meters. Substrate composition varies from solid rock to rocky cobble/sand/silt composition with rocky cobble comprising the major fraction of the substrate at most locations (usually $>$ 80 percent).

The oxygen content of the Holston River is usually greater than 5.0 mg/l but may vary as much as 8.0 mg/l diurnally during summer months. Water quality parameters characterisitic of the study area during 1978 are given in Table 1.

During the growing season (March-October), the study area is colonized by aquatic macrophytes including sago pondweed (*Potamogeton pectinatus* L.), American pondweed (*P. nodosus* Poir.), curlyleaf pondweed (*P. crispus* L.), water stargrass (*Heteranthera dubia*, Jacquin, MacM.), eel grass (*Vallisneria americana* Michx.), Canadian elodea (*Elodea canadensis* Michx.), and the aquatic mosses (*Fissidens fontanus* (B-Pyl.) Steud. and *Leptodictyum riparium* (Hedw.) Watnst).

Macrophyte Productivity

In this study incremental change in biomass through the growing season was determined to estimate annual net primary production using the assumptions and model of Fisher and Carpenter (1976). This method was selected because extensive cropping occurs in the study area because of large daily variations in stream flow controlled by the upstream power generation facility, and the fact that the Fisher and Carpenter model includes an estimate of mortality prior to maximum biomass and net production after maximum biomass. Both of these values are important when estimating annual net productivity in systems experiencing extensive cropping.

Five sampling stations were selected at areas encompassing morphological variations in the 51 kilometer study area. These stations were selected based on interpretations of low altitude aerial photographs (color infrared) taken in 1977 and field inspections during 1979. Each station was permanently marked (10 × 10 cm posts or lead marker weights) and a 20 m × 30 m sampling plot selected. The plot was graphically divided into 600 potential 1 square meter sampling points. Thirty 0.1 m^2 quadrants were sampled monthly at each station from April through August. The sampling points were randomly selected and located in the field by means of a vector board and meter tape. Sampling consisted of removing by hand all macrophytes (including roots) rooted within a square metal frame having an area of 0.1 m^2. The 0.1 m^2 samples were placed on ice and returned to the lab where each sample was washed and separated into roots and stems by species. The samples were then dried for 24 hours at 105°C, and individual weights of roots and stems for each species in the sample were determined. No point was sampled more than once.

Monthly biomass (g/m^2 DW) and 95 percent confidence limits were calculated for each station and the monthy average biomass (g/m^2 DW) for the Holston River above Cherokee Reservoir was calculated by pooling the samples from the five stations and multiplying the pooled average by the areal coverage of aquatic macrophytes (65 percent or 323.18 ha) reported for this reach of river by EPA (1978).

RESULTS

The average monthly biomass and 95 percent confidence limits for each of the five stations are given in Table 2. Stations 1, 2, and 3 reached peak biomass prior to July 1, while stations 4 and 5 reached peak biomass after mid-July. This is because stations 1-3 were dominated by sago pondweed (a plant that exhibits rapid growth during the early part of the growing season) whereas stations 4 and 5 were dominated by species such as eel grass, water stargrass, and Canadian elodea that reach their maximum growth rate later in the season. Maximum average biomass ranged from 620 g/m^2 at station 2 to 447 g/m^2 at station 3. Station 5 demonstrated the highest production rate, 15.75 $g/m^2/day$ (between June 19 and July 19) while the maximum rates for the other stations ranged from 8.25 $g/m^2/day$ at station 1 to 12.43 $g/m^2/day$ at station 2. To estimate the monthly average river biomass, the data for each of the five stations pooled and the average biomass and 95 percent confidence limits calculated (Table 3). Peak biomass was reached by mid-July (427 g/m^2) and the maximum rate of production (7.6 $g/m^2/day$) was between mid-May and mid-June.

These data were then used to construct an annual biomass curve (in this study the senescence portion of

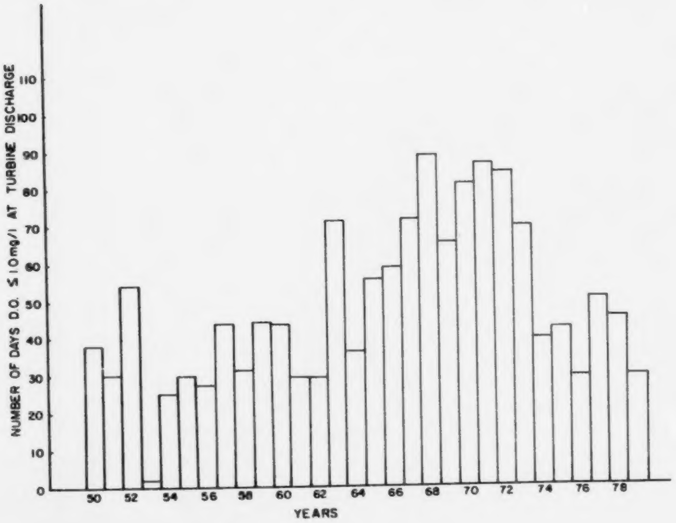
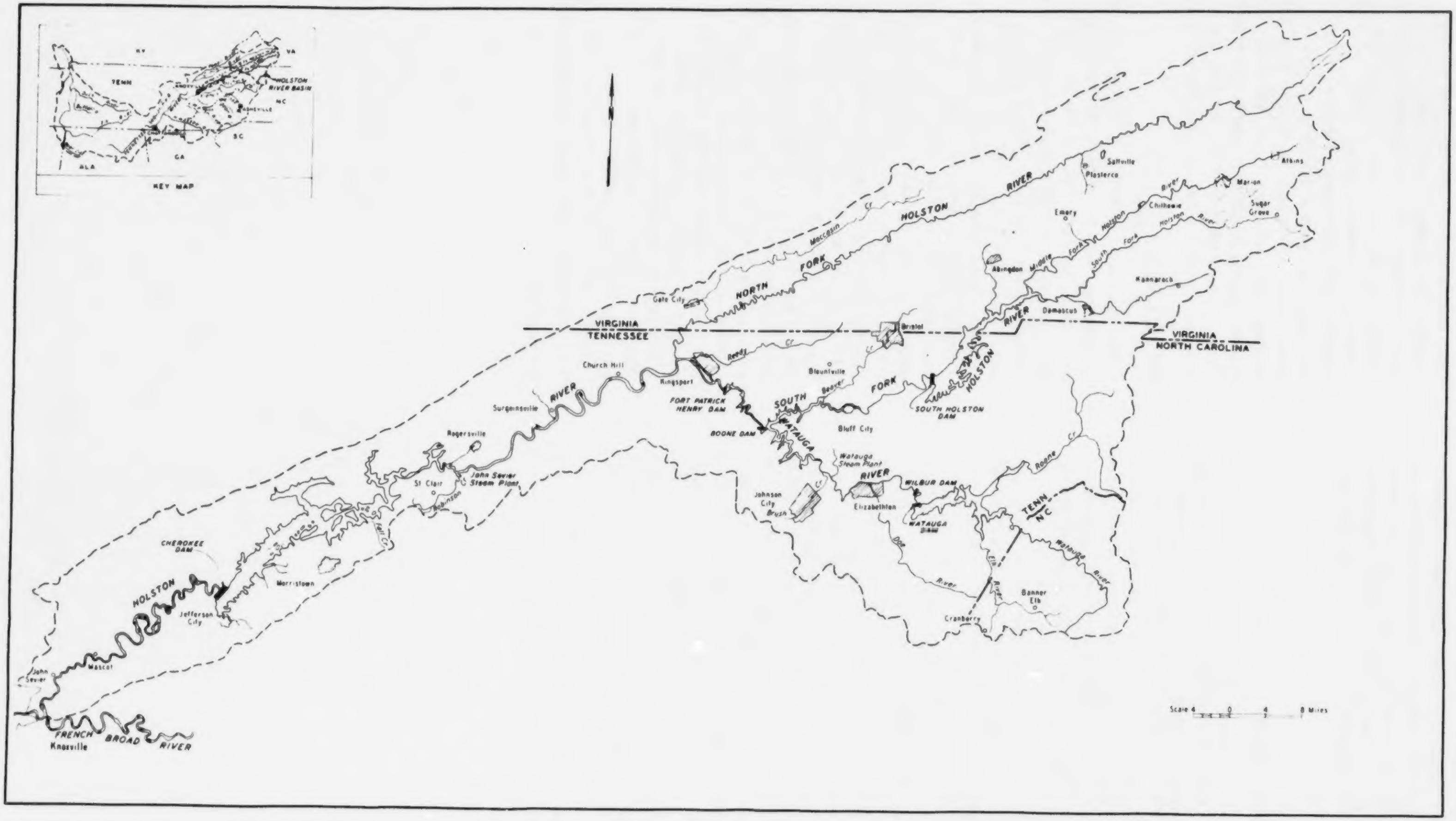


Figure 1. — Location of the study area.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TENN
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Marion
Sugar Grove
Emery
Chilhowie
Middle Fork Holston River
South Fork Holston River
Kannarock
Abingdon
Damascus
HOLSTON RIVER
NORTH FORK
Moccasin Cr
Gate City
VIRGINIA
TENNESSEE
VIRGINIA
NORTH CAROLINA
Bristol
Reedy Cr
Blountville
Beaver Cr
SOUTH FORK HOLSTON RIVER
SOUTH HOLSTON DAM
Bluff City
Kingsport
FORT PATRICK HENRY DAM
BOONE DAM
WATAUGA RIVER
Watauga Steam Plant
WILBUR DAM
WATAUGA DAM
Elizabethton
Johnson City
Brush Cr
Roane Cr
TENN
NC
Doe River
Elk River
Watauga River
Banner Elk
Cranberry
Church Hill
Surgoinsville
RIVER
Rogersville
St Clair
John Sevier Steam Plant
Robinson
Fall Cr
CHEROKEE DAM
Morristown
HOLSTON
Jefferson City
Mascot
John Sevier
FRENCH BROAD RIVER
Knoxville
Scale 4 0 4 8 Miles

Figure 2. — Historical trend in dissolved oxygen cocentrations mg/l from Cherokee Dam.

the curve, i.e., September-December, was estimated based on biomass data for station 3 taken during the previous fall and winter). The biomass curve was converted to a rate curve from which the average annual net productivity was calculated using the procedure of Fisher and Carpenter (1976).

Using this approach, the cumulative annual net production for aquatic macrophytes in the 51 kilometer reach of the Holston River above the Cherokee pool is estimated to be 16.60 metric tons/hectare/year. This value is 3.9 times the maximum biomass of 4.27 metric tons/ha. This indicates that on an annual basis, the Holston River contributes nearly four times its peak biomass of aquatic macrophytes to Cherokee Reservoir. This turnover rate is greater than the estimate of 1 to 3 times the peak biomass suggested by Westlake (1963) for submersed aquatic macrophyte communites but falls within the 0.5 to 5.0 range reported by Rich, et al. (1971). Considering that the Holston River is nutrient rich, has clear, shallow waters, and is subjected to extensive cropping because of fluctuating flows and velocities associated with operation of the hydroelectric facility at Fort Patrick Henry Dam, a turnover rate of 3.9 crops/year seems reasonable.

Annual macrophyte net productivity (16.6 metric tons/ha/yr) converted to total river biomass (16.6 metric tons/ha/yr × 323.18 ha) is 5,365 metric tons dry weight. EPA (1978) reported plants from the Holston River contained 2.06 percent nitrogen and 0.22 percent phosphorus on a dry weight basis. Assuming these percentages, the average total nitrogen and total phosphorus bound by aquatic plants annually is 110 metric tons and 12 metric tons, respectively. Total organic carbon is estimated to be 2,143 metric tons (AFDW being about 85 percent of DW and organic carbon being 47 percent of AFDW (Westlake, 1966). The potential chemical oxygen demand is estimated to be 5,365 metric tons assuming that 1 gram DW of plant material equals 1 gram BOD_5 (Jewell, 1971).

Discussions

The annual net primary production (16.6 metric tons/ha/yr) of the Holston River above Cherokee Reservoir greatly exceeds the 6 metric tons/ha/yr value for freshwater submersed macrophytes in temperate climates suggested by Westlake (1963) and more closely approximates the 17 metric tons/ha/yr value given for freshwater submerged macrophytes in a tropical system. The average maximum biomass (427 g/m^2) agrees with the data of Peltier and Welch (1968) who reported an average maximum biomass of 457 g/m^2 and 420 g/m^2 for two stations located in the current study area. However, Peltier and Welch (1968) estimated that the areal plant coverage was only 20 percent. EPA (1972, 1978) estimated the maximum biomass to be 155 and 200 metric tons, respectively, for the same reach of the Holston River. This is approximately an order of magnitude less than the 1,378 metric tons (4.27 metric tons/ha × 323.18 ha) average maximum biomass reported in this study.

Jewell (1971) reported the average rate of decay of aquatic weeds to be on the order of 0.086/day at 18°C (variation 0.05 to 0.19). The average time of travel between the head of the study reach (HRM 141.1) and Cherokee pool is 2 to 4 days during the growing season depending on the flow from the North Fork of the Holston River and power generation schedules at Fort Patrick Henry Dam (Ruane and Krenkel, 1978). Therefore, plant material lost to cropping and floating unrestricted from the head of the reach would be reduced by 17 to 34 percent prior to entering the Cherokee pool. This factor would represent a maximum since those plants further downstream would experience shorter times of travel and consequently, would undergo less decay before reaching Cherokee Reservoir.

Extensive decay of plant structures within the river system does occur when plants become impinged on stumps, tree limbs, bridge pilings, etc. No data were collected to quantify the amount of plant material impinged within the river, but from observations, it is believed to amount to only a small percentage of that floating in the river at a given time. Dennis (1976) reports that 88 percent of the detritus in the water column trapped by 0.1 millimeter mesh screens was within 20 centimeters of the surface, indicating that 15 to 30 cm/sec velocities are sufficient to keep the plant material suspended. Therefore, a majority of the detached aquatic plants probably reach Cherokee pool without undergoing significant decay.

Previous studies strongly suggest that inputs of allochthonous organic material originating in the highly productive aquatic macrophage beds in the Holston River below Kingsport, Tenn., adversely impact hypolimnetic DO concentrations in Cherokee Reservoir. The present study indicates that the annual average net primary production of aquatic plants in the Holston River above Cherokee Reservoir is 16.6 metric tons/ha/yr, a rate much higher than reported for rivers in the temperate regions of North America. Assuming 15 percent of the plant material is lost to impingement and decay, the biomass contribution from this reach of river is estimated at 4,570 metric tons dry weight (5,365 × 0.85) to Cherokee Reservoir annually. Deposition, subsequent decomposition, and nutrient release from this large amount of allochthonous aquatic macrophyte input represents 94 metric tons nitrogen, 10 metric tons phosphorus, 4,570 metric tons biochemical oxygen demand, all significant factors in hypolimnetic oxygen depletion in Cherokee Reservoir.

Table 1. — Water quality characteristics of the Holston River above Cherokee Reservoir (1978).

Parameter	Average	Minimum	Maximum
pH (units)	7.6	7.3	8.5
Temperature (°C)	15.1	3.3	23.9
DO (mg/l)	9.2	5.4	13.4
BOD_5 (mg/l)	2.2	1.0	3.6
Turbidity (JTU)	9.0	3.0	20.0
Specific conductance (μmohs at 20°C)	276.0	200.0	370.0
Alkalinity (mg/l $CaCO_3$)	84.0	72.0	100.0
Nitrogen, $NO_2 + NO_3$ (mg/l)	0.79	0.60	1.00
Nitrogen, $NH_3 + NH_4$ (mg/l)	0.09	0.02	0.30
Nitrogen, organic (mg/l)	0.17	0.06	0.36
Phosphorus, total (mg/l)	0.06	0.04	0.10
Calcium, total (mg/l)	33.0	28.0	40.0
Magnesium, total (mg/l)	7.5	6.3	9.1
Sodium, total (mg/l)	12.3	7.7	19.0
Potassium, total (mg/l)	1.7	1.4	2.3
Sulfate, dissolved (mg/l)	28.0	18.0	37.0
Chloride, dissolved (mg/l)	19.0	10.0	31.0
Total dissolved residue (mg/l)	162.0	140.0	220.0

Table 2. — Average monthly biomass and 95 percent confidence limits for aquatic macrophytes at each station* (g/m^2 DW).

Station	April 17	May 13	June 18	July 12
1	12 ± 5	210 ± 78	453 ± 249	321 ± 118
2	17 ± 6	160 ± 54	620 ± 370	374 ± 139
3	60 ± 23	97 ± 36	447 ± 181	380 ± 150
4	240 ± 88	76 ± 24	432 ± 164	491 ± 182
5	2 ± 1	54 ± 21	227 ± 84	605 ± 225

* Large variations in 95 percent confidence limits due to the natural structure of the macrophyte beds which sometimes resulted in samples having no rooted plants.

Table 3. — Average biomass and 95 percent limits of aquatic macrophytes in the Holston River above Cherokee Reservoir (pooled data).

Date	Mean biomass (g/m^2 DW)	95% confidence limits Upper	Lower
4-17-80	66	77	55
5-13-80	119	139	100
6-18-80	400	475	326
7-12-80	427	495	358

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LAKE RESTORATION METHODS DEVELOPED AND USED IN SWEDEN

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ABSTRACT

Lake restoration research has been carried out at the Institute of Limnology, University of Lund since 1966. Lakes damaged by excessive eutrophication, acidification, and lowering of water levels, are the objects of research. In many eutrophicated lakes, phosphorous reduction either by diversion or treatment of effluents did not improve water quality and oxygen conditions as expected. This continuing phosphorus concentration was caused by intensive nutrient recycling from the lake sediments. Lake Trummen, Sweden, improved permanently following removal of the upper nutrient-rich sediment by suction dredging. In another Swedish lake the sediments were oxidized by induced denitrification. An apparatus for injecting chemicals into the lake sediments will be used in a pilot project to convert humic acids in the sediments to sodium humates with ionic exchange properties. Techniques were developed to restore lakes damaged by lowering of the water level and overgrown by dense reeds. The commercially available Limno device was developed to improve lakes' receiving efficiency. Systems combining wastewater treatment and biomanipulation were developed.

INTRODUCTION

In the last century especially, water bodies close to densely populated areas or near intensively used rural areas have shown dramatic changes in water quality.

Although initial experiments in lake restoration were carried out early in this century with Naumann publishing in 1915 results of the restoration of Berlin's Lietzensee, the need to restore lakes first became clear in the mid-20th century when industry seriously began to affect the environment.

In Sweden, practically all lakes close to settlements were highly polluted because they were used as receivers. After improved wastewater treatment methods made it possible to reduce nutrient input, people wanted these environments restored for recreational purposes. The diversion of sewage water frequently did not immediately improve conditions except in lakes which were not excessively eutrophic. The storage of sapropelic mud on top of sediments deposited during oligotrophic conditions usually delayed response. Methods therefore had to be developed to restore the sediment function of unpolluted lakes; that is, to act as nutrient sinks and to recycle as few nutrients as possible.

Not only these polluted lakes were objects of restoration. In a large number of lakes, the water level had been lowered to increase fertile areas for agricultural purposes. After brief usage, most of these drained areas did not produce good crops and were abandoned. However, the lakes, or whatever was left of them, were in many cases irreversibly damaged. Macrophytes such as reeds and sedges had overgrown large parts of these lakes. Even when attempts were made to raise the water level, often the root felt of these reedbeds was lifted to the surface because of intense methane production beneath them. Even for these lakes restoration methods had to be developed.

A third attack on the Swedish environment was first sustained in the last decade, when it became evident that more and more lakes were being damaged by acidification processes induced by the excessive burning of sulfur-containing fuels. These damages to the lake ecosystems are probably the first indications of large scale processes in the soils, possibly leading to decreased forest production. Acidification problems are, of course, not controllable by simple restoration techniques. However, since large numbers of animals and fish usually found in these normally oligotrophic and dystrophic lakes have already vanished, measures had to be taken to save some of these sensitive environments.

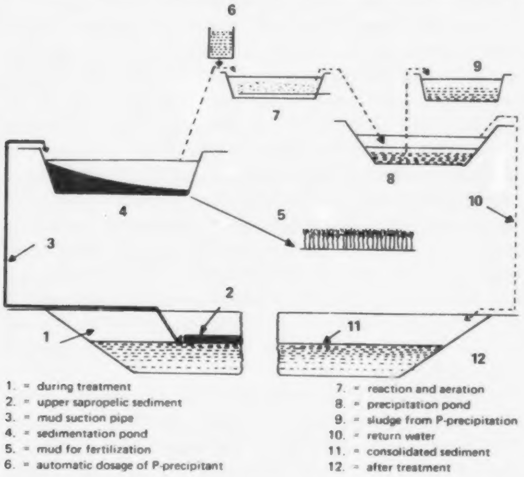
During the last 15 years, a group of workers at the Institute of Limnology in Lund led by Professor S. Bjork have developed restoration methods and carried out restoration activities. They have cooperated closely with technicians to develop new equipment and with authorities to obtain tailor-made solutions for certain lake ecosystems in densely populated areas. Another purpose of this work was to study specific entities of aquatic ecosystems by evaluating the responses of the systems after restoration.

THE RESTORATION OF LAKE TRUMMEN

Lake Trummen, situated in the South Swedish uplands close to the town of Vaxjo, was polluted by municipal sewage and effluents from a textile industry for a period of less than 50 years. When the sewage was diverted to the next lake in the lake system it was expected that this lake which had a relatively short water renewal time of about 3 months, would recover quickly. This was not the case, and after a period of more than 10 years with heavy algal blooms and frequent fish kills it was decided to restore the lake. The preinvestigations showed that the upper 50 centimeters of the sediment were deposited during the pollution period (Digerfeldt, 1972).

The restoration plan provided for removing the upper sediment layer containing the excessive amounts of nutrients deposited during the pollution period. The restoration was carried out during the summer months of 1970 and 1971; about 300,000 m^3 mud were removed. The sediment was deposited in dewatering ponds and the backwater from these ponds was reduced in phosphorus by a small treatment plant with P-precipitation (Figure 1).

The conditions in the lake improved practically instantaneously. Microcystis, dominant before restoration from early spring until autumn, vanished and nanoplanktic species appeared (Gelin and Ripl, 1978; Cronberg, 1980). Nutrient concentrations decreased drastically and good oxygen conditions were maintained during the whole year (Bjork, et al. 1979). The lake is now used for recreational purposes such as fishing and bathing. The experiences from restoring Lake Trummen showed clearly that the internal processes were controlled by microbially mediated exchange processes at the sediment water interface.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9
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2
1
11
12
1. = during treatment
2. = upper sapropelic sediment
3. = mud suction pipe
4. = sedimentation pond
5. = mud for fertilization
6. = automatic dosage of P-precipitant
7. = reaction and aeration
8. = precipitation pond
9. = sludge from P-precipitation
10. = return water
11. = consolidated sediment
12. = after treatment

Figure 1. — Principal scheme for treatment of Lake Trummen (Sweden).

THE BIOCHEMICAL OXIDATION OF THE SEDIMENT IN SITU

As experiments with different sediment cores of various properties showed, release of phosphorus was mainly enhanced in reduced sediments loaded with fresh organic material because of the intense microbial activity exerted by anaerobic bacteria. The intensity of the bacterial processes was shown to be a function of the quality of the organic substance, the presence of a suitable electron acceptor, and temperature. Experiments where nitrate was added to the sediments as an electron acceptor showed that it was possible to oxidize not only easily degradable organic matter by induced denitrification, but also to oxidize the inorganic environment as sulfur and iron species. By this oxidation the sediment became again phosphorus-sorbing and the phosphorus concentrations in the sediment interstitial water decreased drastically.

A tentative restoration was carried out in spring 1975 in a small Swedish lake, Lake Lillesjon close to Varnamo. A harrow to distribute the chemicals was developed in cooperation with the Atlas Copco Co. (Figures 2 and 3). The restoration procedure took 3 weeks. Three chemicals, 13 tons $FeCl_3$ (146 g Fe/m^2), 5 tons slaked lime (180 g Ca/m^2), and 12 tons $Ca(NO_3)_2$ (141 g N/m^2) were injected to an area of 1.2 hectares of reducing sediments. All nitrate was denitrified during 1.5 months. Since that time the lake has stabilized at lowered trophic conditions. The internal nutrient recycling with respect to phosphorus and nitrogen decreased immediately to only 1/6 of the original values. Dense duckweed development during summer stratification over the whole surface of this 4.2-hectare lake was replaced by phytoplankton with summer transparencies of 1.5 to 2.5 meters (Ripl, 1976, 1978).

After this tentative treatment, restoration measures were planned for the Lake Trekanten in Stockholm (Ripl and Lundquist, 1977). This restoration was carried out in May 1980 with a newly designed application harrow (Figure 4). The method is now offered commercially by

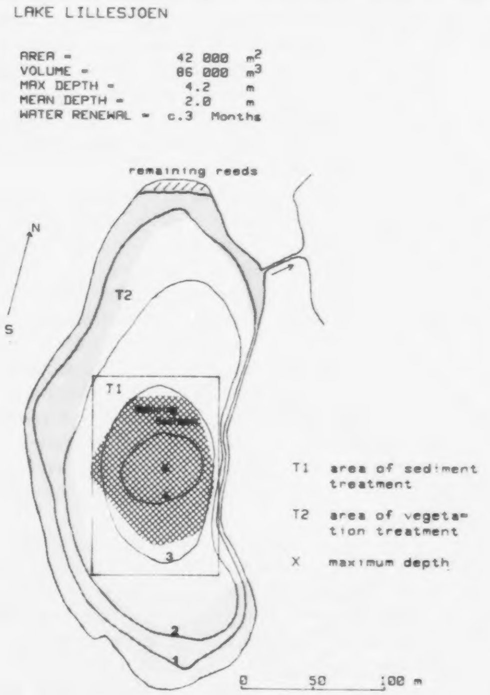

LAKE LILLESJOEN
AREA = 42 000 m2
VOLUME = 86 000 m3
MAX DEPTH = 4.2 m
MEAN DEPTH = 2.0 m
WATER RENEWAL = c.3 Months
remaining reeds
N
S
T2
T1
3
2
1
T1 area of sediment treatment
T2 area of vegetation treatment
X maximum depth
0 50 100 m

Figure 2. — Map showing treated and morphometric data of Lake Lillesjoen (Sweden).

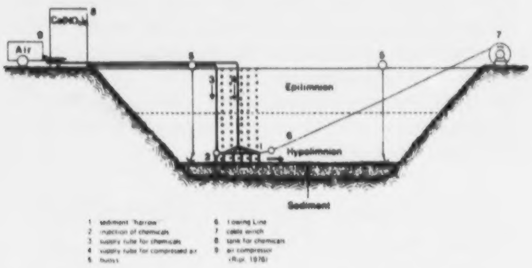


Figure 3. — Principal scheme for the RIPLOX treatment.

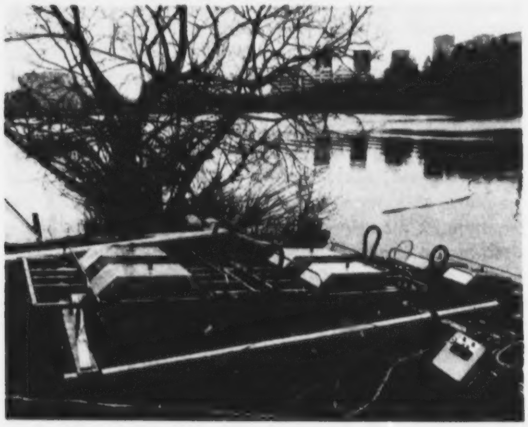


Figure 4. — The sediment harrow has three sections and a total width of 10 meters. Each section has five rows of supply tubes. The three rows in the middle inject air to suspend the sediments. The rear row (depending on direction of movement) applies the chemicals.

Atlas Copco under the trademark "Riplox". So far the results from this recent restoration of Lake Trekanten seem to be the same as in the tentative treatment. Unlike Lake Lillesjon whose sediments were very low in iron compounds because of prolonged periods of anoxic conditions, Lake Trekanten had plenty of natural iron compounds, especially iron sulfides, present in its sediments; injection of iron and lime was therefore unnecessary at Trekanten.

The induced denitrification process was obtained after a short lag period (about 1 week) and at the end of July 1980, 70 percent of the injected nitrate had been denitrified. The oxygen demand of the sediments had reduced drastically; oxygen was found during the summer stratification until the end of July in the whole hypolimnetic zone. The sediments had become phosphorus sorbing and the phosphate concentrations in the interstitial water had decreased from 2 to 4 mg PO_4-P/l to values between 0.01 and 0.3 mg/l in the most reducing sediments at maximum depth. Despite the high denitrification activity followed by vigorous emanation of gas from the sediments, the stratification was preserved and the water in the euphotic zone was never reached by nitrate concentrations higher than about 0.5 to 1 mg NO_3-N/l.

The lake has, of course, not stabilized yet and will probably be labile with respect to planktonic and fish populations. About 20 to 25 centimeters of the upper sediment layers have been oxidized by this induced denitrification process, enabling the benthic fauna to recolonize large areas of the sediments and the lake ecosystem to reach a new steady state with improved loading conditions. The total costs for the restoration of Lake Trekanten were $170,000 or $1.3 per m^2.

THE RESTORATION OF ACIDIFIED LAKES

Many lakes in large areas of Scandinavia, as well as Canadian lakes, are suffering from excessive loading with hydrogen ions, produced by the extensive use of fossil fuel containing sulfur compounds. These acid rains have already partially sterilized thousands of lakes. Until now the only measures that have been taken to save some sensitive fish and crayfish species were liming; however, acid precipitation has caused most humic substances in these lakes to sediment. The addition of lime instantly increases pH values, if this lime is applied from the lake surface in the form of calcium hydroxide. But after a short period the effect of lime is reduced because calcium humates precipitate from the water and the lime reacts with humic substances in the sediments to become insoluble calcium humates (Figures 5 and 6). Another way of lime inactivation is to coat lime particles with humic substances; this reduces the potential for neutralizing acid rain.

In laboratory experiments these effects were investigated and it could be shown that injecting soda solutions directly to the sediments neutralizes the acidic groups, and the sodium humates which are partially soluble react like ionic exchange resins. The gradually introduced acidic rain just exchanges the sodium ions. This treatment is about five to seven times as efficient as adding lime on an equivalent basis. This means that although the chemicals are about three times as expensive, the treatment is more long lasting.

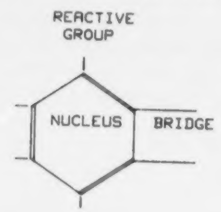
The preinvestigations showed that this sediment treatment with soda probably will be competitive with lime treatment when the longer lasting effect and chemical costs are considered. But an even more pronounced positive effect of the soda-sediment treatment is the natural aluminum phosphate precipitation of extremely nutrient impoverished lakes. The increased exchange processes between water and sediment, after the sediments have been treated with soda, not only increase alkalinity, but also nutrients, leading to primary production and a self-maintaining recycling of nutrients. A certain primary production is a prerequisite for maintaining a fish population.

Another advantage of this treatment is that the pH is not as affected as with frequently conducted liming measures, thus producing more stable physical-chemical conditions suitable for the populations characteristic of ecosystems.

A tentative treatment in the acidified Lake Lilla Galtsjon in Blekinge, Sweden will be carried out this year by the restoration team in Lund, Atlas Copco, and the author. The pretreatment studies in this lake were mainly concerned with evaluating the sediment

THE CONTRACID METHOD

STRUCTURE OF HUMIC ACIDS



REACTIVE GROUPS:

ACIDIC

ALKALINE

DISSOCIATION

$-OH \longrightarrow O^- H^+$

$=NH \longrightarrow N^- H^+$

$-COOH \longrightarrow COO^- H^+$

Figure 5. — Structure of humic aids.

THE CONTRACID METHOD

REACTION OF HUMIC ACIDS WITH

1 LIME PRODUCTS	2 SODA PRODUCTS
$Ca(OH)_2$	$NaOH$
CaO	Na_2CO_3
$CaCO_3$	

R= HUMIC RESIDUE

$R-O-Ca-$	$R-O^- \_\_ Na^+$
$R=N-Ca-$	$R=N^- \_\_ Na^+$
$R-COO-Ca-$	$R-COO^- \_\_ Na^+$
	REVERSIBLE IONIC-EXCHANGE
INSOLUBLE PRODUCTS	SOLUBLE PRODUCTS

Figure 6. — Reaction of humic acids with lime and soda products.

properties, the diffusion of various soda solutions into the sediment, the optimal area to be treated, and the extent of the eutrophication caused by nutrient exchange between sediment and water.

Our purpose is to obtain the most favorable and stable conditions combined with a long lasting effect. The procedure will be suited for strongly acidified lakes and will be available as "Contracid" method. The necessary restoration parameters, however, have to be evaluated in advance by experimental and limnological field work (Ripl, 1978).

THE RESTORATION OF OVERGROWN LAKES AND WETLANDS

A considerable number of lakes in Sweden were damaged by lowering the water level and the resulting expansion of macrophytes. Some lakes which were of great importance for the reproduction of water fowl, or important stations for migrant birds such as cranes, are now restoration objects. One of considerable size — and the largest restoration project in Sweden — is the famous Lake Hornborga. Since it is not possible to fill overgrown lakes again with water without first preparing the lake area which had been overgrown by reed and sedge vegetation, methods had to be developed for such restoration. The usually very resistant root felts had to be cut and removed by amphibious machines and large amounts of accumulated biomass had to be removed and burned; this was done mainly during winter when the ice cover made the use of heavier machines possible. The water level could then be increased, leading almost instantly to the development of underwater vegetation.

The restoration plan for Lake Hornborga includes raising the water level to a maximum depth of 2.4 meters. It should take only one spring to fill the lake with water, as Lake Hornborga is flooded every year after snow melt. The project goal for Lake Hornborga is restoration of an open water area of 11 km². In 1977 the Swedish government decided to spend about $7 million on this restoration (Bjork et al. 1979) (Figure 7).

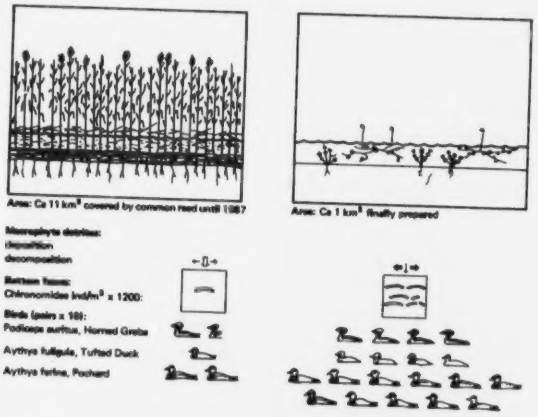

Area: Ca 11 km² covered by common reed until 1967
Area: Ca 1 km² finally prepared
Macrophyte detritus:
deposition
decomposition
Bottom fauna:
Chironomidae ind/m² x 1200:
Birds (pairs x 10):
Podiceps auritus, Horned Grebe
Aythya fuligula, Tufted Duck
Aythya ferina, Pochard

Figure 7. — Lake Hornborga before and after experiments for directing the primary production from emergent to submerged vegetation. Water level not yet raised. Comparison between conditions in 1965 and 1971 (Bjork, 1972).

Draining large areas in Sweden for agricultural purposes has destroyed many wetlands. New ecological insight has led to projects to restore wetlands. The reestablishment of wetlands in drained lakes and peat pits implies the production of energy reeds, potential habitats for waterfowl, fish, and wildlife, and shallow water reservoirs for the recessive amphibious fauna (Bjork and Graneli, 1978).

AERATION METHODS

Eutrophic lakes of a certain depth, especially receivers of polluted effluents, suffer during stagnation periods of insufficient oxygenation of the hypolimnetic zone. Furthermore, lagoons used for storage and the breakdown of a heavy load of industrial oxygen-demanding effluents prior to their discharge need additional oxygen. In drinking water reservoirs the addition of oxygen will prevent the dissolution of iron and manganese compounds from the bottom areas and thereby avoid the relatively expensive water treatment in a treatment plant. For this reason different aeration devices have been developed and used. In Sweden the original idea proposed by Bernhardt and Hotter (1967) to achieve aeration with an air lift was further developed by Atlas Copco and resulted in the Limno device. Thorough limnological studies in connection with aeration measures showed that it is possible to control oxygen abundance and thereby improve aerobical breakdown of autochthonous and allochthonous organic material in aquatic environments still overloaded by organic matter, or by excessive nutrients.

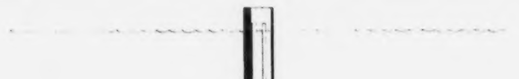


Figure 8. — Cutaway sketch of polyester plastic limno unit.

Examples for the application of these aeration devices are now numerous in Sweden and abroad. One of Sweden's mining companies has installed five Limno units with a capacity of about 250 kg O /day each in several basins receiving effluents from an ore flotation process using organic flotation chemicals. Another example is the emergency drinking water reservoir for the town of Brussels where a Limno unit was installed to prevent anaerobic conditions. Other lakes are maintained by aeration until final solutions to divert pollutants such as phosphorus precipitation are installed. In West Berlin one of the Havel lakes, the Tegeler See, used by thousands of people for swimming and other recreational purposes is equipped with three Limno units; it will eventually be equipped with eight or nine more units, each delivering 350 kg O_2/day, until the phosphorus elimination plant is built (Figure 8).

BIOMANIPULATION AND OPTIMIZATION OF THE COMPLEX TREATMENT PLANT RECEIVER

Simultaneous to the development of the different restoration methods, changes in structure and function of the treated ecosystems were analyzed. Andersson (1977) investigated the relationships between phytoplankton, zooplankton, and various species of fish. He was able to show in both limnocoral experiments and whole lake studies that the various fish species by selecting their diet and thereby controlling the abundance of filter feeders had a more or less eutrophicating effect. Selective fishing in Lake Trummen for roach and bream resulted in increased transparency, lowered nutrient level, and reduced biomass. Experiments were simultaneously conducted in enclosures in this lake. The results from these experiments showed even more pronounced effects (Andersson, 1979).

Ripl (1978) proposed experiments to show that a nitrification step in the advanced sewage treatment plants with phosphorus reduction, and a direction of the plant effluents to the reducing sediment areas would oxidize the reducing sediments, denitrify excessive nitrogen, and increase the phosphorus-sorbing capacity of the sediments. Part of the nitrified effluents would, of course, serve as a nitrogen source for algae and probably induce a succession of nitrogen-fixing blue-green algae. These experiments were partially carried out in 1978 (Ripl, et al. 1979) and showed that most of the added nitrate nitrogen was denitrified in the sediments of even shallow systems. Further, the almost monoculture of the nitrogen-fixing species *Anabaena flos aquae* broke down and was replaced by green algae. Transparency increased since the green algal population was controlled by filter feeders. There are now plans for some receiver-ecosystems to try this concept in Lake Finjasjon in Scania, Sweden, and in the large German fjord Schlei (Figure 9).

Theoretical knowledge of aquatic ecosystems is growing stronger at the limnological institutes. This means that the management of lake ecosystems as well as their maintenance and in some cases their restoration becomes safer. However, since lake ecosystems are individuals, there will never be a best method which can be applied to every lake ecosystem. Only knowledge of the structure and function of each individual system makes it possible to develop suitable methods to reach a new equilibrium with changed and improved conditions.

Figure 9a. —

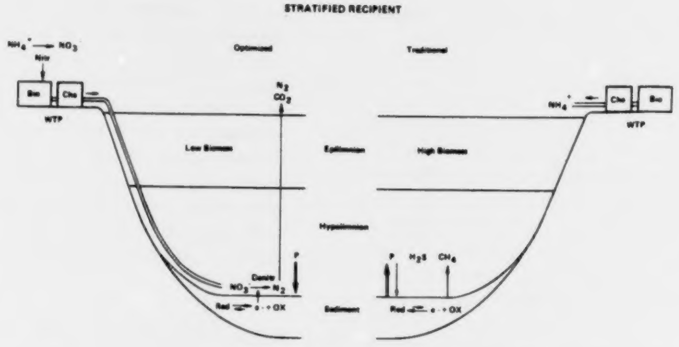

STRATIFIED RECIPIENT
$NH_4^+ \rightarrow NO_3$
Nitr
Optimized
Traditional
Bio
Che
WTP
N_2
CO_2
NH_4^+
Che
Bio
WTP
Low Biomass
Epilimnion
High Biomass
Hypolimnion
P
Denitr
$NO_3^- \rightarrow N_2$
Red ⇌ e⁻ + OX
Sediment
P
H_2S
CH_4
Red ⇌ e⁻ + OX

Figure 9b. —

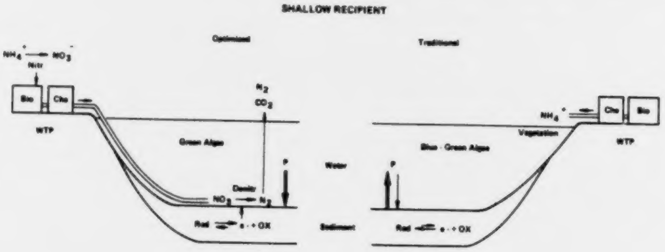

SHALLOW RECIPIENT
Optimized
Traditional
$NH_4^+ \rightarrow NO_3^-$
Nitr
Bio
Che
WTP
N_2
CO_2
NH_4^+
Che
Bio
WTP
Vegetation
Green Algae
Blue - Green Algae
P
Water
P
Denitr
$NO_3^- \rightarrow N_2$
Red ⇌ e⁻ + OX
Sediment
Red ⇌ e⁻ + OX

Figure 9a, b. — Schematic diagrams of optimized and traditional models of the treatment plant/recipient system (stratified recipient). WTP = wastewater treatment plant with biological treatment (Bio) and chemical precipitation (Che). Nitr = nitrification process. Denitr = denitrification process. (Ripl, et al. 1979).

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SUMMARY OF CLEAN LAKES PROJECTS

Name: Albert Lea
Location: Freeborn County, Minn.

Problem: High phosphorus content, depleted D.O., turbid water, and severe algal bloom.

Project Objectives: To restore the water quality and to restore recreation activities.

Restorative Techniques Used: Relocate treatment plant effluent by implementing a 201 project. Filter stormwater runoff through native soil filter media. The feasibility of dredging and other in-lake activities will be determined following diversion of the effluent.

Project Progress: Fountain Lake soil filters have been completed; Albert Lea Lake 201 project has been approved.

Implementation Problems: None.

Name: Allentown
Location: Allentown Borough, N.J.

Problem: Pollution inputs have resulted in a hypereutrophic condition which is exemplified by nuisance algal blooms and aquatic weeds plus sedimentation.

Project Objectives: Reduce pollutant inputs from both the watershed and the lake sediments; remove sediments.

Restorative Techniques Used: Dredge sediments, stabilize the shoreline, divert stormwater pollution.

Project Progress: Project implementation expected to begin in 1981.

Implementation Problems: Delays caused by lengthy 404 permit process.

Name: Ann Lee
Location: Albany County, N.Y.

Problem: Nuisance macrophytes, accelerated eutrophication, shallowness, runoff from agricultural, residential, and commercial areas.

Project Objectives: Remove accumulated sediments and reduce sediment and nutrient inputs.

Restorative Techniques Used: Dredge sediments; treat runoff, and stabilize shoreline.

Project Progress: Dredging is underway; construction of retention pond in design phase.

Implementation Problems: None.

Name: Apopka
Location: Orange and Lake Counties, Fla.

Problem: Fish kills, algal blooms, water hyacinths, hypereutrophication.

Project Objectives: Restore water quality of lake by reducing internal nutrient loading from unconsolidated sediments.

Restorative Techniques Used: Drawdown lake to consolidate bottom sedimentation and reduce internal nutrient loading.

Project Progress: Preliminary monitoring and engineering design study have been completed as has an Environmental Impact Statement.

Implementation Problems: Costs of the project, due to environmental constraints, turned out to be prohibitively expensive. Project was terminated in 1980.

Name: Ballinger
Location: King-Snohomish Counties, Wash.

Problem: Excessive algal and macrophyte growths and turbidity affecting swimming, boating, and picnicking.

Project Objectives: Control external sediment/nutrient sources and reduce in-lake nutrient buildup.

Restorative Techniques Used: Tributary sedimentation basins; reduce lake level fluctuations; remove polluted hypolimnetic water.

Project Progress: Sedimentation basins complete; monitoring and evaluation continuing.

Implementation Problems: Unclear responsibility for lake level control; interjurisdictional disagreement on lake project operation and impacts.

Name: Bantam
Location: Litchfield County, Conn.

Problem: Lakewide phytoplankton blooms and macrophyte beds in the extensive littoral zones which cover as much as 20 percent of the lake's 371 surface hectares.

Project Objectives: To deepen the lake and reduce aquatic macrophytes, and to improve recreational opportunities.

Restorative Techniques Used: Selective dredging of 343,970 cu. meters of sediment from those areas where sufficient organic sediment exists to promote growth of aquatic macrophytes; nonpoint source loading abatement program for the lake watershed.

Project Progress: Watershed study work plan development is in progress.

Implementation Problems: None.

Name: Big Alum
Location: Worcester County, Mass.

Problem: Septic tank leachate from shoreline residences and erosion are causing high concentrations of phosphorus in the lake, pointing to eventual eutrophic conditions.

Project Objectives: Preserve the present water quality of the lake.

Restorative Techniques Used: Develop plan which may include: sedimentation basins, composting toilets, and modified septic systems; public participation and education program; watershed management; purchase and management of wetlands areas.

Project Progress: Engineering design study is underway and should be completed in spring 1981.

Implementation Problems: Numerous delays in awarding contract.

Name: Blue
Location: Monona County, Iowa

Problem: Heavy siltation; low water levels; dense growth of macrophytes; and decreasing lake usage.

Project Objectives: Restore water quality and deepen lake.

Restorative Techniques Used: Dredge approximately 36 percent of lake to remove aquatic vegetation and nutrient-enriched bottom sediment; form sanitary district to ensure proper construction and operation of treatment facilities in the future.

Project Progress: Dredging has been completed; water quality assessment underway.

Implementation Problems: None.

Name: Bomoseen
Location: Rutland County, Vt.

Problem: High nutrient concentrations have resulted in heavy growth of aquatic macrophytes and blue-green algae, which interfere with recreational activities.

Project Objectives: Since nutrient sources entering the lake have been controlled, the goal is to remove the in-lake nutrient source, which is recycling from existing aquatic vegetation.

Restorative Techniques Used: Harvesting 73 hectares of the lake each year for 3 years will remove excessive nutrient levels, thereby reducing aquatic plant growth and increasing public access and use of the lake.

Project Progress: 3 years of harvesting has been completed. To date, harvesting has limited plant growth as indicated by the decrease in pounds of aquatic macrophytes removed from harvested areas. Final project assessment is now underway.

Implementation Problems: None.

Name: Broadway
Location: Anderson County, S.C.

Problem: Siltation

Project Objectives: Remove lake sediments and reduce erosion and sedimentation in the watershed above the lake.

Restorative Techniques Used: Best management practices, dredging, and roadbank stabilization.

Project Progress: Roadbank stabilization in progress; archeological study in progress; pre-monitoring program complete.

Implementation Problems: Dam safety issue has prevented release of funds for sediment dredging. The COE will not commit to dredging in the lake until the dam is repaired and passes a safety inspection.

Name: Buckingham
Location: Albany County, N.Y.

Problem: Extensive aquatic plant growth; lake bottom covered with organic debris and silt.

Project Objectives: Improve overall lake water quality.

Restorative Techniques Used: Drain lake and remove the accumulated silt and muck by excavation.

Project Progress: Removal of accumulated sediment did not substantially reduce the eutrophic state of the lake.

Implementation Problems: None.

Name: Bugle
Location: Trempealeau County, Wis.

Problem: Sediment infilling has reduced the recreational value of the lake.

Project Objectives: To remove most of the sediment in the lake and to enhance streambank stabilization upstream.

Restorative Techniques Used: Hydraulic dredging, riprapping, and limited sloping and seeding.

Project Progress: Project is 70 percent complete. Some dredging and watershed work will have to be carried over to summer 1981.

Implementation Problems: Heavy rains in August 1980, plus farmers' reluctance to allow heavy machinery into their fields slowed the project.

Name: Charles River
Location: Suffolk County, Mass.

Problem: Saltwater stratification prevents vertical mixing, and decomposition of organic materials results in complete oxygen depletion in the deeper zones resulting in odor from hydrogen sulfide production.

Project Objectives: Destratify the basin and improve water quality.

Restorative Techniques Used: Destratification of the Charles River lower basin by induced circulation using compressed air.

Project Progress: Austin equipment in place. Destratification has been effective. Final water quality assessment is performed.

Implementation Problems: None.

Name: City Park Lakes
Location: East Baton Rouge Parish, La.

Problem: Heavy metals sedimentation, hypereutrophication and agricultural runoff.

Project Objectives: Restore lakes' water quality by reducing nutrient inputs and sediment removal.

Restorative Techniques Used: Dredging to remove sediments; control of urban stormwater runoff; rehabilitation of sewer lines; institute agricultural BMP's; and divert water to aid in faster stormwater dissipation.

Project Progress: Completed preliminary feasibility work and are ready to bid the dredging work.

Implementation Problems: Multiple coordination has caused project delays.

Name: Clear
Location: Waseca County, Minn.

Problem: High nutrient content and algal bloom in summer.

Project Objectives: Restore water quality of Clear Lake.

Restorative Techniques Used: Diversion of stormwater through a marsh filter system.

Project Progress: Approximately 60 percent completed.

Implementation Problems: Delays have been caused by adverse weather.

Name: Clearwater River Chain of Lakes (Clearwater, Augusta, Caroline, Marie, Louisa, Scott, Betsy Lakes)
Location: Wright, Stearns, and Meeker Counties, Minn.

Problem: Excessive nutrient loading, algal blooms, and proliferation of macrophytes.

Project Objectives: Restore the recreational, aesthetic, and water quality of the Chain of Lakes.

Restorative Techniques Used: Wetland treatment systems for stormwater runoff from tributary streams will be used. On-land disposal of three community effluents is being done now. Hypolimnetic alum treatment of Augusta Lake will be undertaken.

Project Progress: Work plan is being finalized.

Implementation Problems: None.

Name: Cobbossee Watershed District I
Location: Kennebec County, Maine

Problem: Three lakes comprising the watershed (Annabessacook, Cobbossee, and Pleasant Pond) are eutrophic and suffer from excessive phosphorus enrichment and dense algal blooms.

Project Objectives: Reduce phosphorus loading to the lakes.

Restorative Techniques Used: Hypolimnetic aeration to control internal nutrient cycling; chemical addition (alum) to bind or absorb soluble phosphorus; construction of manure storage facilities to control phosphorus runoff; diversion of runoff; and livestock exclusion from streams.

Project Progress: Watershed nutrient controls have been almost completed. In-lake work has finished. Monitoring is ongoing to assess the project results.

Implementation Problems: Reluctance of farmers to initially participate in the program because of unproven benefits.

Name: Cobbossee Watershed District II
Location: Kennebec County, Maine

Problem: A dozen lakes in the Cobbossee watershed are being threatened by agricultural runoff.

Project Objectives: Protect these lakes in this watershed by implementing agricultural BMP's.

Restorative Techniques Used: Implement agricultural BMP's to reduce nutrient inputs to lakes. Control measures will include manure storage facilities, diversion of barnyard runoff.

Project Progress: The work plan has been finalized and implementation is awaiting assessment of agricultural BMP's from the Cobbossee I project.

Implementation Problems: None.

Name: Cochituate
Location: Suffolk County, Mass.

Problem: Eutrophication is resulting in excessive blue-green algal production, odor problems, oxygen depletion, and possible loss of cold water fishery.

Project Objectives: Reduce influx of nutrients from surface water runoff and septic tank seepage.

Restorative Techniques Used: Purification of tributary water by natural sand filter beds; dredging of three settling ponds and installation of an automatic nutrient inactivation system in first settling pond; public awareness program; drawdown; and harvesting of rough fish for nutrient removal.

Project Progress: Filter beds have been evaluated; nutrient budgets have been computed; and a technical memorandum on the methodologies, costs, and impact of dredging has been completed. Reassessment of restoration alternatives is underway.

Implementation Problems: Cost effectiveness of project has been questioned. More engineering work needs to be done.

Name: Cochrane
Location: Duel County, S.D.

Problem: Blue-green algal blooms caused by nutrient influx from agricultural runoff.

Project Objectives: Reduce nutrient input to lake.

Restorative Techniques Used: Construct three sediment control dams to intercept runoff and construct settling basins behind the dams to catch sediments and nutrients.

Project Progress: Sediment traps have been developed and preliminary evidence suggests that the influx of suspended solids has been greatly reduced.

Implementation Problems: None.

Name: Collins Park
Location: Schenectady County, N.Y.

Problem: Sediment and nutrient loadings from a storm sewer outfall are causing sediment buildup and aquatic vegetation growths.

Project Objectives: Reduce sediment and nutrient loadings so lake might be used for recreational activities.

Restorative Techniques Used: Dredging; planting of macrophytes to act as a nutrient trap; and removal of snow and cut vegetation (which were dumped in or near the lake) from the lake drainage area.

Project Progress: Dredging has been completed.

Implementation Problems: None.

Name: Commonwealth
Location: Washington County, Ore.

Problem: Siltation and excessive algal growths preventing use of the lake.

Project Objectives: Identify silt and nutrient sources; develop and implement corrective measures.

Restorative Techniques Used: Dredging, dilution, riprap, and revegetation.

Project Progress: Project completed—achieved greater depth and clarity of lake water. Developed attractive lake setting and facility for fishing, boating, and picnicking.

Implementation Problems: None.

Name: Covell
Location: Minnehaha County, S.D.

Problem: Eutrophication and sediment loading to the lake.

Project Objectives: Improve water quality, develop better fisheries habitat.

Restorative Techniques Used: Dredging, modification of outlet structure, and construction of sediment retention pond.

Project Progress: Engineering design completed and dredging to begin shortly.

Implementation Problems: 404/402 permit requirements for discharge of dredge elutriate have delayed the project.

Name: Creve Coeur
Location: St. Louis County, Mo.

Problem: Sedimentation is resulting in decreasing surface area and depth.

Project Objectives: Increase surface area and depth of Creve Coeur Lake and improve recreational opportunities.

Restorative Techniques Used: Dredge 121 hectares to a depth of 3 meters; dredged spoils are to be deposited in the area surrounding the lake and used for lake development.

Project Progress: Dredging to begin in early 1981.

Implementation Problems: Coordination problems between region and locals have caused several delays in project implementation.

Name: Decorah
Location: Juneau County, Wis.

Problem: Excessive sediment.

Project Objectives: Remove and transport sediment and improve water quality.

Restorative Techniques Used: Hydraulic and mechanical dredging.

Project Progress: Work plan has been completed, but project work has not been implemented.

Implementation Problems: A lawsuit challenging the legality of the formation of the lake district is in court. Until this is resolved, no work is being done.

Name: Delaware Park
Location: Erie County, N.Y.

Problem: Floating debris, siltation, and sewage deposits from Scajaquada Creek resulted in the lake being closed for public use.

Project Objectives: Reduce and/or remove pollution entering the lake from Scajaquada Creek.

Restorative Techniques: Install stormwater interceptors; detour Scajaquada Creek around the lake through a closed underground conduit; finally, dewater and dredge the lake. Refill the lake with clean spring water.

Project Progress: Stream diversion conduit has been completed. Dewatering and dredging are scheduled to begin in 1981.

Implementation Problems: None.

Name: Ellis
Location: Yuba County, Calif.

Problem: Urban runoff has caused excessive nutrients and sedimentation and growths of the nuisance macrophyte, *Hydrilla.*

Project Objectives: To rehabilitate lake by eliminating excessive growths of hydrilla, and diverting stormwater flow from the lake.

Restorative Techniques Used: Nonpoint source control program, removal of sediments, application of herbicides.

Project Progress: Sediments have been removed and stormwater interceptor system has been constructed.

Implementation Problems: Project costs have been high.

Name: Ellis Brett Pond
Location: Plymouth County, Mass.

Problem: Pond is eutrophic and nonpoint source pollution including stormwater runoff from a regional shopping center has made the pond unsafe for swimming.

Project Objectives: Reduce impact of nonpoint source pollution and remove accumulated sediments and problem aquatic plants.

Restorative Techniques Used: Streetsweeping; installation of filters and oil traps on parking lot drains; construction of catch basins; and dredging.

Project Progress: Engineering study showed costs for dredging and catch basins would be extremely expensive.

Implementation Problems: Project cancelled because it was judged not to be cost effective by grantee.

Name: Eola
Location: Orange County, Fla.

Problem: Urban runoff resulting in hypereutrophication.

Project Objectives: Reduce urban runoff into the lake and restore it for increased public usage.

Restorative Techniques Used: Parking lot diversion into percolation ponds; street inlet modification for percolation; and diversion of runoff through natural areas.

Project Progress: Water quality monitoring program in progress; designs for the percolation and lake natural systems are complete; pilot percolation basin completed; construction to begin in early 1981.

Implementation Problems: None

Name: Lake Fenwick
Location: King County, Wash.

Problem: Turbidity and sediment interfering with park development and lake use for boating, fishing, and swimming.

Project Objectives: Control stormwater and erosion of banks and bottom of inlet stream.

Restorative Techniques Used: Enforce clearing and grading ordinance; divert peak flow of inlet stream; and provide detention for inlet stream water.

Project Progress: Clearing and grading ordinance being enforced; inlet stream diversion system complete; detention basin and outlet stream to lake complete; revegetation plan complete and work scheduled; monitoring and evaluation continuing.

Implementation Problems: Inability to obtain easement for better diversion pipeline route resulting in in-stream pipeline.

Name: 59th Street Pond
Location: New York, N.Y.

Problem: The pond is stagnant and turbid with excessive growths of algae and grasses; substantial reduction of water depth from siltation; high color levels; and high coliform content.

Project Objectives: Restore quality of pond to increase its value as a passive recreational source for tourists and local residents.

Restorative Techniques Used: The pond will be drained and dredged. The bottom of the pond will be made impenetrable to prevent remaining bottom nutrients from entering the water column. Pond bank riprap will be repaired, and clogged stormwater drainage pipes will be cleaned.

Project Progress: Construction and dredging are completed. Post-construction monitoring is underway. Preliminary results indicate a marked improvement in the appearance of the pond.

Implementation Problems: The total project costs have been high ($250,000/acre).

Name: Finger Lakes (12 lakes)
Location: Boone County, Mo.

Problem: All of the lakes are acidic as a result of acid mine drainage caused by exposed sulfurous spoil areas.

Project Objectives: Improve water quality of the lakes by eliminating acid sources.

Restorative Techniques Used: Connect 12 separate lakes by construction of five small earthen dams and two canals to form a single lake of 17 hectares; divert to project lakes the drainage of 405-hectare rural watershed not disturbed by mining.

Project Progress: Construction completed and final assessment underway.

Implementation Problems: None.

Name: Frank Holten Lakes
Location: St. Clair County, Ill.

Problem: Accumulated silt deposits and nutrients caused by runoff have degraded water quality of all three lakes.

Project Objectives: Restoration of lakes to suitable depth and rehabilitation of fish population. Relocation of Harding Ditch, a major source of pollutants.

Restorative Techniques Used: Dredging, relocation of Harding Ditch, and construction of inverted siphon.

Project Progress: Dredging projects for lakes 1 and 2 are expected to be in engineering design during the winter of 1980-81.

Implementation Problems: Administrative delay and budgetary problems necessitated an extension of the project and budget periods.

Name: Gibralter
Location: Santa Barbara County, Calif.

Problem: Lake is filling in with sediments. Some of the sediment is contaminated with mercury from past mining activities.

Project Objectives: Remove contaminated sediments.

Restorative Techniques Used: Dredge sediments using a "Pneuma" pump method.

Project Progress: Planning completed. Dredging should begin in winter 1980.

Implementation Problems: Delays in planning due to project complexity. Dispute about patents and rights to the Pneuma pump.

Name: Green Valley
Location: Union County, Iowa

Problem: Shallowness, excessive sedimentation and runoff.

Project Objectives: Reduce runoff and sedimentation and deepen the lake.

Restorative Techniques Used: Dredging to remove accumulated sediments, agricultural BMP's.

Project Progress: Work plan being developed.

Implementation Problems: None.

Name: Half Moon
Location: Eau Claire County, Wis.

Problem: High phosphorus loading is a major cause of abundant nuisance algae and indirectly creates excessive oxygen demands with resultant fish winterkill.

Project Objectives: To reduce phosphorus loading.

Restorative Techniques Used: Storm sewer diversion and installation of supplemental wells.

Project Progress: Storm sewer work is underway, collectors are being installed, and project is 80 percent complete.

Implementation Problems: The makeup wells did not provide sufficient water. Collectors are being extended from the less porous ground near the wells to the gravels associated with the nearby Chippewa River.

Name: Hampton Manor
Location: Rensselaer County, N.Y.

Problem: The lake has a eutrophic condition with algal blooms in summer months, and the encroachment of rooted macrophytes is threatening recreational activities.

Project Objectives: Restore lake by removing sediments; oxygenate the bottom waters.

Restorative Techniques Used: Drawdown of lake; consolidation and removal (dredging) of sediments. Placement of aeration system.

Project Progress: Turbidity and algal blooms have been reduced; transparency has been improved.

Implementation Problems: None.

Name: Lake Harriet/Lake of the Isles
Location: Hennepin County, Minn.

Problem: Urban stormwater runoff.

Project Objectives: To improve the water quality of Lake Harriet and Lake of the Isles.

Restorative Techniques Used: Vacuum sweep streets that drain into Lake Harriet, and install first flush diverters in the Lake of the Isles drainage area.

Project Progress: Diverters are completed and are monitored on a storm basis. Vacuum sweeping continuing on project.

Implementation Problems: Difficulties in scheduling street vacuuming.

Name: Henry
Location: Trempealeau County, Wis.

Problem: Excessive sedimentation.

Project Objectives: Increase water depth, reduce sedimentation, and reduce nutrient inflow.

Restorative Techniques Used: Hydraulic dredging, streambank stabilization by rock riprapping, sloping and seeding, and selective fencing.

Project Progress: Project completed. The lake was dredged, streambank stabilization was achieved, and runoff from barnyards upstream was diverted. Project assessment is underway.

Implementation Problems: None.

Name: Herman
Location: Lake County, S.D.

Problem: Advanced eutrophication, algal blooms, low D.O., occasional fish kills.

Project Objectives: Reduce sediment and nutrient loadings to lake.

Restorative Techniques Used: BMP's and sediment control structures in the watershed.

Project Progress: The project is half complete. BMP's and sediment control structures are almost in place. Plans are being formulated for additional in-lake restorative work.

Implementation Problems: None.

Name: Hyde Park
Location: Niagara County, N.Y.

Problem: Deteriorating quality due to increased pollution loading from housing developments, a sanitary landfill, accidental oil spills from a railroad yard, and sedimentation.

Project Objectives: Improve overall quality of lake by reducing pollutant loadings and removing sediment.

Restorative Techniques Used: Drain and dredge lake; augment flow to lake; plant native vegetation along streambank to retard erosion; construct siltation pond; install oil boom system downstream from siltation pond; and carry out limnological monitoring program.

Project Progress: Watershed measures are being implemented including sewering 900 homes, proper landfill management, and control of pollutants from the railway yard. Dredging is underway and should be completed in 1981. Sedimentation pond construction is underway.

Implementation Problems: None.

Name: Hyland
Location: Hennepin County, Minn.

Problem: High phosphorus content, algal blooms, and turbid water.

Project Objectives: Restoration of lake water quality.

Restorative Techniques Used: Lake drawdown, treat bottom sediments for phosphorus removal, build stormwater settling ponds, and drill wells for flow augmentation.

Project Progress: All implementation work completed.

Implementation Problems: After lake was drained, an enormous growth of smartweed had to be harvested so that nutrients would not be reintroduced during flooding.

Name: Jackson
Location: Leon County, Fla.

Problem: Nonpoint source pollution, sediment and nutrient loading into the lake.

Project Objectives: To reduce sediment and nutrient load entering the lake from nonpoint sources.

Restorative Techniques Used: A filtration impoundment system coupled to a marsh to reduce nutrient and sediment loading.

Project Progress: All land purchased, lagoons and marsh filtration system is underway.

Implementation Problems: Land acquisition has been delayed several times by high appraised values but all property has been purchased.

Name: Kampeska
Location: Codington County, S.D.

Problem: Shoreline erosion and high sediment loading.

Project Objectives: Reduce shoreline erosion and control input of nutrients to the lake.

Restorative Techniques Used: Riprapping shoreline areas.

Project Progress: Riprapping is near completion and sediment loading rates have been developed.

Implementation Problems: One of the riprap areas failed due to the steep slope. The slope could not be modified prior to riprapping due to its historic nature.

Name: Lafayette
Location: Alameda and Contra Costa Counties, Calif.

Problem: Excessive growth of blue-green and other algal types creates taste and odor problems and clogs the filter of the nearly completed water treatment plant; low oxygen concentration in the hypolimnion.

Project Objectives: Restore the recreational, aesthetic, and economic values of Lafayette Reservoir.

Restorative Techniques Used: Hypolimnetic aerations and nutrient inactivation.

Project Progress: Project has not been implemented. Only water quality monitoring program was undertaken.

Implementation Problems: Project was not implemented because of cost increases and problems securing additional local funds.

Name: Lansing
Location: Ingham County, Mich.

Problem: The shallowness of the lake has allowed for extensive macrophyte growth and has resulted in recreational impairment.

Project Objectives: To restore recreational use, especially boating, and to improve aesthetics and fish population.

Restorative Techniques Used: Hydraulic dredging of lake bottom, and beach nourishment through depositing of dredged sand on selected beaches.

Project Progress: More than 30 percent of dredging has been done.

Implementation Problems: Implementation has been hindered by controversial actions from the Township to the Federal level. Delays have been caused by court battles. The delays have contributed to cost increases, including the general factor of inflation and the particular factor of greatly increased fuel costs.

Name: Lenox Reservoir
Location: Taylor County, Iowa

Problem: Eutrophic, highly turbid with odor and taste problems; extensive siltation; increasing macrophyte growth.

Project Objectives: Restore overall water quality of Lenox Reservoir by deepening the lake and removing vegetation.

Restorative Techniques Used: Dredging and some dike construction to insure that dredged material does not return to the lake.

Project Progress: Project has been completed and water quality goals have been accomplished.

Implementation Problems: None.

Name: Liberty
Location: Spokane County, Wash.

Problem: Excessive blue-green algal growth reducing boating, swimming, and aesthetic values.

Project Objectives: Reduce external and internal nutrient sources; inactivate phosphorus and provide sediment release barrier.

Restorative Techniques Used: Discontinue septic tanks implementing 201 program, marsh water manipulation/diversion, selective dredging, alum sulfate treatment, stormwater management program.

Project Progress: Marsh water control completed; alum treatment-dredging underway; monitoring and evaluation continuing.

Implementation Problems: Coordination with State game department; locating dredge spoils disposal area.

Name: Lilly
Location: Kenosha County, Wis.

Problem: In-filling with accumulated organic materials, rough fish, and reduced recreational opportunities.

Project Objectives: Restore lake fisheries and deepen to prohibit winter fish kills.

Restorative Techniques Used: Dredging with cutterhead hydraulic dredge.

Project Progress: Dredging completed, dredging equipment removed, booster pumps removed, and dikes around spoils removed. Final landscaping will be finished in 1981.

Implementation Problems: Wet summer during 1980 prevented final spoil incorporation into the soils and landscaping.

Name: Little Muskego
Location: Waukesha County, Wis.

Problem: Severe infilling and rooted emergents in the near-shore area affect approximately 40 percent of the lake.

Project Objectives: Increase recreational opportunities and improve water quality.

Restorative Techniques Used: Dredging the lake.

Project Progress: Preliminary studies completed.

Implementation Problems: Potential arsenic contamination of local groundwater supplies; State-required EIS; local dissent to proposed disposal sites; and spiraling costs.

Name: Little Pond
Location: Lincoln County, Maine

Problem: Heavy growth of zooplankton was causing taste and odor problems in water distribution lines.

Project Objectives: Alleviate taste and odor problems.

Restorative Techniques Used: Introduce alewives to control zooplankton population.

Project Progress: Plankton populations were reduced and potability of Little Pond water was increased. Project was successful.

Implementation Problems: None.

Name: Loch Raven
Location: Baltimore County, Md.

Problem: Excessive seasonal algal blooms, and high manganese levels during every fall reservoir turnover.

Project Objectives: Insure potable water in the Baltimore area is of high quality and free from objectionable tastes and odors.

Restorative Techniques Used: Install a diffusive aeration system for the purpose of destratifying the reservoir.

Project Progress: Monitoring of reservoir water quality. Workplan development to assess different aeration systems including wind-driven aerators and hypolimnetic aeration is presently underway.

Implementation Problems: Numerous procedural delays in carrying out the project; all bids for installing aeration system significantly exceeded the budgeted amounts.

Name: Lone Star
Location: Douglas County, Kans.

Problem: Shallowness, excessive sedimentation, poor water quality.

Project Objectives: Deepen lake and reduce sedimentation.

Restorative Techniques Used: Control erosion by shoreline stabilization; dredge to remove excessive sediments.

Project Progress: Work plan being developed.

Implementation Problems: None.

Name: Long
Location: Kitsap County, Wash.

Problem: Excessive algal and weed growths interfering with boating, swimming, and fishing.

Project Objectives: Reduce external and internal sources; inactivate phosphorus and provide sediment release barrier.

Restorative Techniques Used: Septic tank zoning and clearing/grading ordinances; outlet area dredging; drawdown and beach renovation; aluminum sulfate treatment.

Project Progress: Considerable improvement in water clarity; beach improvement completed (private beach owners) and macrophyte reduction achieved. Monitoring and evaluation continuing.

Implementation Problems: Inability to obtain dredging contractor delayed project 1 year; dredge temporarily shut down due to high disposal area turbidity.

Name: Long Lake Chain of Lakes
Location: Ramsey County, Minn.

Problem: High phosphorus content in Chain of Lakes, algal blooms severe, turbid water during storms, and stormwater runoff.

Project Objectives: Prevent, remove, reduce, and eliminate pollution of Long Lake Chain of Lakes.

Restorative Techniques Used: Sedimentation basins, channel repairs, upstream BMP's, wetlands treatment systems, and dredging.

Project Progress: Total project is approximately 65 percent completed. Sedimentation basins, channel repairs, and wetland treatment systems have been constructed. Dredging has not been started.

Implementation Problems: Keeping contractors on schedule because of delays caused by weather conditions.

Name: Lower Mystic
Location: Suffolk County, Mass.

Problem: Construction of a dam in 1909 resulted in the entrapment of 946 million liters of saltwater in two deep kettle holes in the lake. The anoxic zone has generated high concentrations of sulfides, ammonia, and phosphorus.

Project Objectives: Remove salt water; aerate bottom waters; and reduce sulfide concentrations.

Restorative Techniques Used: Pump saline water from the lake; remove hydrogen sulfide by precipitation with ferric chloride; and aerate bottom waters.

Project Progress: Work plan has been completed and construction has begun.

Implementation Problems: None.

Name: Manawa
Location: Pottawattamie County, Iowa

Problem: Excessive sedimentation and aquatic macrophyte growth.

Project Objectives: Improve water quality, deepen the lake, and improve fishing.

Restorative Techniques Used: Dredging to remove accumulated sediments.

Project Progress: Work plan has been accepted and dredging is scheduled to start in late 1980.

Implementation Problems: None.

Name: Marinuka
Location: Trempealeau County, Wis.

Problem: Excessive sedimentation with consequent large growth of nuisance aquatic plants.

Project Objectives: Removal of sediments.

Restorative Techniques Used: Dredge 653,937 cubic meters of sediment and stabilize upstream banks by riprap, sloping, seeding, and fencing critical areas.

Project Progress: Work plan is being developed.

Implementation Problems: Leakage in the lake's dam requires a study currently underway to determine what repairs are necessary.

Name: McQueeney
Location: Guadalupe County, Tex.

Problem: Sedimentation; heavy macrophyte growth;

algal blooms; and occasional anaerobic conditions in the hypolimnion.

Project Objectives: Restore water quality by removing sediments.

Restorative Techniques Used: Dredging to increase depth of the lake and to prevent macrophyte growth.

Project Progress: Dredging is about to begin.

Implementation Problems: Project delay due to bid for dredging being $100,000 over planned amount.

Name: Medical
Location: Spokane County, Wash.

Problem: Excessive blue-green algae and low dissolved oxygen preventing fish survival, boating, and swimming.

Project Objectives: Inactivate phosphorus and provide a sediment release barrier.

Restorative Techniques Used: Aluminum sulfate treatment.

Project Progress: Phosphorus and chlorophyl are considerably reduced and blue-green algae under control. Lake returned to high level of boating, water skiing, swimming, picnicking, and fishing. Report available.

Implementation Problems: Wind effect on alum distribution barges. Impurities in liquid alum supply.

Name: Mirror/Shadow
Location: Waupaca County, Wis.

Problem: Advanced eutrophication due to stormwater has caused algal blooms, high phosphorus concentrations, and fish winterkills.

Project Objectives: Divert stormwater discharges, immobilize phosphorus in the bottom sediments, and increase winter dissolved oxygen concentrations.

Restorative Techniques Used: Construction of new storm sewers away from lake; application of aluminum sulfite to precipitate phosphorus and to seal bottom; and installation of aeration system in Mirror Lake.

Project Progress: Project completed. External phosphorus loading rates were reduced 65 percent, and internal phosphorus rates were also reduced. Aeration has increased dissolved oxygen.

Implementation Problems: None.

Name: Moore
Location: Ramsey County, Minn.

Problem: Moore Lake is a shallow, eutrophic lake maintained primarily by stormwater runoff. It has 1 meter of organic muck on top of a firm clay bottom.

Project Objectives: Halt the eutrophication of Moore Lake.

Restorative Techniques Used: Elimination of external phosphorus sources by diversion or treatment; inactivation of nutrient in sediments; and dredging of sediment deltas.

Project Progress: Work plan is being developed.

Implementation Problems: None.

Name: Morse Pond
Location: Norfolk County, Mass.

Problem: High nutrient loading from urban runoff and sediments has resulted in blue-green algal blooms, and high organic loading from deciduous leaves has resulted in color problems.

Project Objectives: Control algae and nutrient and organic loadings.

Restorative Techniques Used: Chemical treatment for iron and colloidal particle removal; harvesting; dredging; public education; and replacing deciduous trees with evergreens.

Project Progress: Seminars have been conducted and newspaper articles written in compliance with the educational program activities. One of two wetland areas around the lake has been purchased as a buffer zone. Chemical treatment has been applied to the lake. All dredging has been completed. Project assessment is underway.

Implementation Problems: None.

Name: Moses
Location: Grant County, Wash.

Problem: Excessive algal growths interfering with boating, swimming, and fishing.

Project Objectives: Identify implementable agricultural BMP's; have sewage treatment plant discharge removed from lake; determine effective lake dilution rates and volumes; implement lake dilution system using Columbia River water.

Restorative Techniques Used: Lake dilution.

Project Progress: Pilot dilution study complete; State EIS complete; agricultural BMP study in progress; monitoring and evaluation continuing.

Implementation Problems: Assurance of permanent availability of dilution water, low State 201 funding for removal of sewage treatment plant discharge from lake.

Name: Mystic
Location: Rutherford County, N.C.

Problem: Use of the lake has been seriously impaired by aquatic weed growth and high turbidity, both caused by increased sedimentation.

Project Objectives: Renovation of the lake to provide recreational opportunities (swimming, boating, fishing, etc.)

Restorative Techniques Used: Dredge existing sediment deposits and use dredged material for the construction of two sediment control dams. Construct spillways and install riprap along the shore.

Project Progress: Construction has been completed. Water quality assessment underway.

Implementation Problems: None.

Name: Noquebay
Location: Marinette County, Wis.

Problem: Excessive aquatic vegetation has greatly reduced open water and impaired recreational value.

Project Objectives: To harvest the aquatic nuisance plants, and to demonstrate whether weed harvesting is a viable technique for removing nutrients that have accumulated in a lake.

Restorative Techniques Used: Mechanical weed harvesting.

Project Progress: Two seasons of harvesting have been completed. There are some indications that harvesting results in the development of less dense but more diverse weed patches the year after treatment. It has not yet been determined if the nutrients in the lake are actually being reduced.

Implementation Problems: Problems in acquiring and maintaining harvesting machines caused considerable delay. There have been problems quantifying the benefits resulting from the project, but two independent studies are underway to accomplish this task.

Name: North Park
Location: Allegheny County, Pa.

Problem: Excessive siltation which has caused a reduction in public usage of the lake.

Project Objectives: The removal of accumulated sediment and the restoration of lake water quality.

Restorative Techniques Used: Dredge approximately 130,787 cubic meters of sediment.

Implementation Problems: Costs of the project have increased significantly and project has had to be scaled down.

Name: Nutting
Location: Middlesex County, Mass.

Problem: High nutrient levels; blue-green algae; low transparency; nuisance aquatic vegetation; high oxygen demand of mucky sediments; color; and organic sediment accumulation.

Project Objectives: Improve overall quality of lake for recreational activities.

Restorative Techniques Used: Dredging and post-dredging flocculation; control of overland runoff inputs by street sweeping, sediment entrapment; establishment of buffer zones; public education; and diversion of stormwater around the lake.

Project Progress: Detailed scope of work, including the identification of dredged material disposal areas and program budget, has been developed. Dredging has begun and will continue for 2 more years.

Implementation Problems: None.

Name: Oakwoods
Location: Brookings County, S.D.

Problem: Sediment loading and unstable shoreline.

Project Objectives: Bank stabilization and improve water quality of the lake.

Restorative Techniques Used: Riprapping of shoreline areas.

Project Progress: Eroding shoreline banks have been stabilized.

Implementation Problems: Archeological site within one riprap area. This caused project delays because it required designation as eligible for National Register and excavation prior to finishing of project.

Name: Oelwein
Location: Fayette County, Iowa

Problem: Excessive siltation and shallowness.

Project Objectives: Improve water quality and deepen lake.

Restorative Techniques Used: Dredge to remove accumulated sediments; construct sedimentation ponds.

Project Progress: Dredging has been completed and siltation ponds have been constructed.

Implementation Problems: None.

Name: Pauls Valley
Location: Garvin County, Okla.

Problem: Excessive sedimentation.

Project Objectives: Reduce sedimentation and restore lake's water quality.

Restorative Techniques Used: Construct flood control structures and erosion control ponds, including BMP's in grass planting, critical area planting, cross fencing, rotational grazing, diversion terraces, field terraces, pasture fertilization.

Project Progress: Work plan is being developed.

Implementation Problems: None.

Name: Penn
Location: Scott County, Minn.

Problem: Dissolved oxygen depletion and urban stormwater runoff over gardens and lawns.

Project Objectives: Aerate lake and divert stormwater.

Restorative Techniques Used: Pump well water over stair-step outlet and introduce to the lake, and build filter ponds at storm outlets.

Project Progress: Work approximately 85 percent completed.

Implementation Problems: None.

Name: Phalen
Location: Ramsey County, Minn.

Problem: High phosphorus content, algal bloom, and stormwater runoff.

Project Objectives: To restore the water quality of the lake.

Restorative Techniques Used: Divert runoff through marsh filter, and address upstream BMP's. Installation of holding ponds for storm sewers.

Project Progress: Progress has been slow and the project, in the final design stage, is only 10 percent complete.

Implementation Problems: As a result of public and neighbors' objecting to in-lake holding pond and bottom sealing, both of which were dropped from work program, construction has not started.

Name: Reeds
Location: Kent County, Mich.

Problem: Eutrophication at an accelerated rate, with filamentous algal blooms and macrophyte growth in the littoral zone in summer, and reduction of "game fish" populations and recreational usefulness.

Project Objectives: To improve water quality.

Restorative Techniques Used: Reduction of phosphate in surface runoff by passage and enforcement of a debris-bagging ordinance and the City's sale of no-phosphate fertilizers.

Project Progress: Cooperation from citizens has been excellent. No construction has taken place.

Implementation Problems: The City signed contracts without EPA approval. A little work (mostly of a monitoring or research nature) was accomplished before the City was asked not to use its letter of credit.

Name: Rivanna Reservoir
Location: Albermarle County, Va.

Problems: Taste and odor problems, fish kills, heavy blooms of blue-green algae, high nutrient loading.

Project Objectives: To calculate the efficiency and cost effectiveness of several nutrient management and lake restorative pilot projects.

Restorative Techniques Used: Installed grassed waterway on crop land; constructed residential sedimentation ponds; installed an aeration system in the reservoir.

Project Progress: Aeration system has been installed and construction of grassed waterways and sedimentation ponds has been completed. Results of the pilot studies are being assessed.

Implementation Problems: None.

Name: Ronkonkoma
Location: Suffolk County, N.Y.

Problem: High coliform bacteria counts and stormwater runoff inputs of nutrients and toxic metals.

Project Objectives: Reduction in coliform bacteria, heavy metal and nutrient inputs. Increase public uses.

Restorative Techniques Used: Diversion of stormwater runoff; installation of biofiltration ponds; shoreline stabilization.

Project Progress: Two ponds have been installed. Preliminary data suggest that the marsh ponds can remove significant amounts of stormwater pollutants.

Implementation Problems: Land acquisition problems have caused significant project implementation delays. These delays have resulted in changes in the scope of the project. Local administrative and management problems have been significant.

Name: Rothwell
Location: Randolph County, Mo.

Problem: Excessive siltation and inputs of nutrients.

Project Objectives: Rehabilitate the lake's silted-in area by removing the accumulated sediments.

Restorative Techniques Used: Dredging to remove sediments.

Project Progress: None.

Implementation Problems: No action taken because locals are having difficulties raising matching funds.

Name: Sebasticook
Location: Penobscot, Maine

Problem: Excessive nutrient loading has led to a condition of hypereutrophy with classical symptoms of chronic dense algal blooms, increased vascular plant growth, and fish kill.

Project Objectives: To improve lake water quality and recreational opportunities.

Restorative Techniques Used: The proposal provides for dam reconstruction in order to permit a 3.5 meter drawdown of the lake. The drawdown in concert with point and nonpoint source controls is expected to significantly improve water quality.

Project Progress: Watershed work plan is being developed. Final negotiation is underway for dredging work.

Implementation Problems: None.

Name: Sabattus Pond
Location: Androscoggin County, Maine

Problem: In recent years, the pond has deteriorated due, in part, to the existence of nuisance blooms over most of the summer. In fact, water contact recreation is severely restricted every summer.

Project Objectives: To improve lake water quality and recreational opportunities.

Restorative Techniques Used: The proposal provides for dam reconstruction and outlets to permit a 3-meter drawdown of the pond. Other work includes dredging and nonpoint source control to improve lake water quality.

Project Progress: Watershed work plan is being developed for agricultural lands.

Implementation Problems: None.

Name: Sacajawea
Location: Cowlitz County, Wash.

Problem: Excessive algal and macrophyte growths and turbidity affecting swimming, boating, fishing, and picnicking.

Project Objectives: Remove external and internal sources of nutrients and dilute with low nutrient river water.

Restorative Techniques Used: Intercept and divert stormwater outfalls; dilute with low nutrient Cowlitz River water; remove nutrient sediment and macrophytes by dredging.

Project Progress: Stormwater diversion system complete; flushing/dilution system under construction; dredging plan in progress; monitoring and evaluation continuing.

Implementation Problems: Mt. St. Helens' mud in Cowlitz River preventing completion of dilution water system and using up dredged material disposal sites in Longview area.

Name: Sacajawea
Location: Park County, Mont.

Problem: Extremely shallow, high sediment and nutrient loading, low in-flow.

Project Objectives: Restore water quality and fish habitat.

Restorative Techniques Used: Sediment removal; diversion of sediment-laden in-flow tributary; flow augmentation.

Project Progress: Bids are being let for construction of the in-flow line. The lake has been drained to allow for sediment excavation.

Implementation Problems: Poor estimates on project's cost required modification and review of project scope.

Name: Salmon
Location: Kennebec, Maine

Problem: Salmon Lake once supported a diverse cold water fishery; however, recently only brown trout were able to maintain themselves. Obvious signs of eutrophication are apparent with noxious algal blooms occurring frequently.

Project Objectives: To improve lake water quality and recreational opportunities.

Restorative Techniques Used: Modification of a dairy farm drainage area. A 3-year construction phase during which diversions, tiles, a storage lagoon, and irrigation system will be built.

Project Progress: Watershed work plan being developed. Project implementation is awaiting results of Cobbossee I project.

Implementation Problems: None.

Name: Scudders Pond
Location: Nassau, N.Y.

Problem: Excessive sedimentation, stormwater runoff, advanced state of eutrophication, blue-green algal blooms.

Project Objectives: Remove excessive sediments; increase public use.

Restorative Techniques Used: Dredge sediments; control incoming sedimentation problem by constructing stormwater retention basins.

Project Progress: Work plan completed, project bid accepted, contract let for dredging.

Implementation Problems: Obtaining local matching funds.

Name: Skinner
Location: Noble County, Ind.

Problem: Sediment and nutrient-contaminated runoff from the lake's agricultural watershed is causing sediment buildup at the stream outlet and weed growth around the shallow edge of the lake.

Project Objectives: Reduce sediment and nutrient runoff in combination with sediment settling and nutrient filtering; remove accumulated sediments and use weed removal and chemical application to eliminate existing weed growth in lake.

Restorative Techniques Used: Control of sediment and agricultural runoff pollutants by conservation practices, channel stabilization, and dredging.

Project Progress: Watershed conservation practices have been implemented. Large sediment basin will be constructed in 1981 and channel stabilization and sediment dredging will also begin in 1981.

Implementation Problems: Local matching funds for large sediment basin, dredging, and channel stabilization were less than originally planned and progress was delayed. Farmers in the project area have rejected nonstructural conservation practices such as reduced tillage and the use of cover crops in favor of structural practices, usually parallel tile outlet terraces.

Name: Spada/Chaplain
Location: Snohomish County, Wash.

Problem: Increased turbidity preventing adequate disinfection of raw water supply serving 200,000 people.

Project Objectives: Identify turbidity sources; develop and select best turbidity control plan; develop and adopt interjurisdictional basin resource management plan.

Restorative Techniques Used: Stream channel modification-riprapping; gabion construction around blue clay outcroppings; selected slope area revegetation; resource management plan.

Project Progress: Project completed—reduced turbidity to acceptable level for simple chlorination.

Implementation Problems: Obtaining agreement among jurisdictions, i.e., County, State (DNR & Health), USFS, and private owners on objectives and management practices.

Name: Stafford
Location: Marin County, Calif.

Problem: Eutrophication as evidenced by algal blooms, high levels of organic matter associated with lake sediment, seasonally high nutrient levels, and high coliform bacteria concentrations.

Project Objectives: Control of organic and nutrient inputs.

Restorative Techniques Used: Dry excavation of sediment from the lake and erosion control. Spoil to be used in expansion of present park area.

Project Progress: Necessary property for buffer zone has been purchased; erosion control has been implemented; removal of sediment will be initiated in late 1980.

Implementation Problems: Project delays due to drought conditions experienced during 1976-1977 that prevented the drawdown of Stafford Lake.

Name: Steinmetz
Location: Schenectady County, N.Y.

Problem: Sediment accumulation, macrophyte problem, accelerated eutrophication.

Project Objectives: Remove sediments, improve water quality.

Restorative Techniques Used: Dredge sediments, place sand on beach areas, divert stormwater runoff.

Project Progress: Project has been completed. Turbidity has decreased, water quality has improved, public usage has increased.

Implementation Problems: None.

Name: Summit
Location: Summit County, Ohio

Problem: Accelerated eutrophication caused by urban nonpoint source runoff.

Project Objectives: Remove nonpoint source nutrient inputs to the lake.

Restorative Techniques Used: Retention and/or diversion of stormwater runoff and in-lake aeration have been tentatively identified as restorative techniques.

Project Progress: Work plan developed to study stormwater runoff abatement practices.

Implementation Problems: None.

Name: Sunset
Location: Texas County, Okla.

Problem: Sedimentation has caused water quality problems.

Project Objectives: Stop rapid sedimentation; restore water quality; improve fish habitat; repair dam and overflow pipe; and dredge lake.

Restorative Techniques Used: Draining and sediment removal; shoreline stabilization by vegetation and soil cement; and construction of upstream impoundments.

Project Progress: Work plan formulation underway. Sensibility study has been implemented.

Implementation Problems: None.

Name: Swan
Location: Turner County, S.D.

Problem: High sediment loading due to shoreline wave erosion.

Project Objectives: Reduce sediment loading and stabilize bank area.

Restorative Techniques Used: Riprapping of shoreline and renovation of outlet structure.

Project Progress: Shoreline areas have been stabilized.

Implementation Problems: Inclement weather caused numerous delays during riprapping and construction.

Name: Sylvan
Location: Custer County, S.D.

Problem: Excessive sedimentation influx.

Project Objectives: Protect lake from future sedimentation; reduce erosion of surrounding areas.

Restorative Techniques Used: Construction of erosion control structures in camping area, re-seeding and modification of parking area to redirect runoff.

Project Progress: Project is being delayed until State legislature authorizes acceptance of Federal money.

Implementation Problems: None to date.

Name: Tahoe
Location: Washoe and Douglas Counties, Nev.

Problem: Development has increased sediment and nutrient loading, causing an increase in primary productivity and algal growth in near shore areas.

Project Objectives: Control sediment and nutrient contributions in critical erosion areas.

Restorative Techniques Used: Erosion control and slope stabilization structures including rock-lined ditches, rock slope protection, gabion walls, and revegetation.

Project Progress: Plans and specifications approved by EPA on July 25, 1980. Project now in construction bid stage. Grant offer dated July 23, 1980. Offer has not yet been accepted by State of Nevada.

Implementation Problems: The award for Kingsbury Grade has not been accepted because of problems with local funding.

Name: Temescal
Location: Alameda County, Calif.

Problem: Nutrients, coliforms, and sediment are main problems.

Project Objectives: Remove sediments and implement nutrient control program.

Restorative Techniques Used: Dredging sediments; install retention pond; implement source control through city regulation of grading.

Project Progress: Sediments have been removed and retention ponds are being installed.

Implementation Problems: Source control program has been hard to institute: lack of resources and reluctance of city to regulate construction; inability to control activities in watershed.

Name: Thurston Lakes (Long, Patterson, Hick & Lois)
Location: Thurston County, Wash.

Problem: Excessive algal and weed growths interfering with boating, swimming, fishing, and picnicking.

Project Objectives: Evaluate and establish storm and on-site wastewater management program, agricultural BMP's, and in-lake restorative needs.

Restorative Techniques Used: On-site wastewater management programs; stormwater management program; agricultural BMP program; in-lake procedures (to be determined).

Project Progress: Initial work plan being developed.

Implementation Problems: None.

Name: Tivoli
Location: Albany County, N.Y.

Problem: Accumulated raw sewage sludge sediments; stormwater runoff; siltation caused by soil erosion; and pollutants from old, broken sewer lines.

Project Objectives: Clean up water ecosystem; stabilize soil; and develop associated ponds and wetlands.

Restorative Techniques Used: Develop shallow water areas and wetland areas upstream to retard stormwater runoff and reduce siltation; drain and excavate main lake to a maximum depth of 8 to 10 feet; regrade banks and vegetate to prevent erosion; redesign and rebuild existing earthen dike and emergency spillway.

Project Progress: Project completed; accumulated sediments have been removed; water quality has been improved as has public usage of the lake.

Implementation Problems: None.

Name: Upper Willow
Location: St. Croix County, Wis.

Problem: Excessive sediment and excessive growth of emergent and submergent plants.

Project Objectives: Removal of sediments.

Restorative Techniques Used: Riprapping, sloping and mulching, seeding, dredging, and installing a sediment trap.

Project Progress: Work plan is being developed.

Implementation Problems: None.

Name: Vancouver
Location: Clark County, Wash.

Problem: Siltation, excessive algal growths, and high coliform preventing summertime use of the lake.

Project Objectives: Control external sediment and other pollution sources; remove sediments; deepen and contour for maximum circulation; provide dilution water system.

Restorative Techniques Used: Watershed 208 water quality management plan; dredging; flushing (dilution) with Columbia River water.

Project Progress: Pilot dredge study complete; NEPA-Wetland study complete; operations plan complete; permits applied for and bid documents being prepared for dredging and flushing channel construction; monitoring and evaluation continuing.

Implementation Problems: Approval of dredge spoils disposal sites; flushing channel design to prevent salmon migration into lake; State hydraulics and Corps of Engineers 404 permits.

Name: Vandalia Reservoir
Location: Pike County, Mo.

Problem: Siltation from stormwater runoff has reduced the storage capacity of Vandalia Reservoir by 50 percent.

Project Objectives: Improvement of lake water quality and restoration of the impoundment to its original capacity.

Restorative Techniques Used: Dredging of 137,195 meters of bottom sediment and construction of sediment catchment basins in the watershed.

Project Progress: Dredging has been completed. Final assessment of water quality is underway.

Implementation Problems: None.

Name: Lake Wapato
Location: Pierce County, Wash.

Problem: Excessive algal and weed growths interfering with swimming, fishing, and other recreational uses.

Project Objectives: Reduce external and internal sources of nutrients and provide low nutrient dilution water.

Restorative Techniques Used: Stormwater detention basin and diversion system; drawdown for weed control and bottom compaction; dilution system using city Cedar River supply.

Project Progress: Dilution experimental study complete; revised plan complete; final design started; monitoring and evaluation continuing.

Implementation Problems: Scheduling drawdown and diversion system construction for least impact on park activities and lake use.

Name: Lake Waramaug
Location: Kent County, Conn.

Problem: The extensive summer and fall blue-green algal blooms in the lake are the most obvious symptoms of the lake's eutrophication problems. Agricultural runoff from barnyards, feedlots, etc., is a major source of the pollution.

Project Objectives: To improve lake water quality and recreational opportunities.

Restorative Techniques Used: Restoration includes implementation of conservation practices; local land use controls; comprehensive information, education, and public participation programs; water quality monitoring; and project coordination.

Project Progress: Watershed work plan development is underway.

Implementation Problems: None.

Name: Washington Park
Location: Albany County, N.Y.

Problem: Increased lake nutrient levels; reduced transparency and lake depth; excessive aquatic weed growth.

Project Objectives: Improve overall lake water quality.

Restorative Techniques Used: Drain lake and remove bottom sediment by dredging.

Project Progress: Post-restoration monitoring shows an improvement in lake transparency and an elimination of aquatic weed growth along the shorelines.

Implementation Problems: None.

Name: Waterford
Location: Anne Arundel County, Md.

Problem: Insufficient storm drainage system causing erosion and water quality problems.

Project Objectives: Improve the water quality of the lake by reducing the bank and shoreline erosion and the resultant siltation and suspended solids problem in the lake.

Restorative Techniques Used: Construction of closed storm drainage system, timber bulkheading, and gabion shores protection.

Project Progress: Work plan has been approved and project construction is about to begin.

Implementation Problems: None.

Name: White Clay
Location: Shawano County, Wis.

Problem: The lake has become eutrophic because of phosphorus loading from animal wastes.

Project Objectives: To reduce phosphorus loading from animal wastes and from cropland runoff.

Restorative Techniques Used: Seventeen barnyard storage facilities were built. Farmers cooperated in spreading animal wastes onto fields when they thawed. Grassed waterways, terraces, diversions, and reduced tillage were instituted.

Project Progress: Project completed. Total phosphorus loading from all sources of animal waste is estimated to have been reduced from 451 kg in 1970 to 342 kg in 1978, a reduction of 25 percent.

Implementation Problems: Landowners had to pay for manure storage facilities and wait for reimbursement, resulting in some reluctance and delay in the early stages.

CLEAN LAKES PHASE II IMPLEMENTATION PROJECTS

Restoration/Preservation Techniques

In-lake Techniques

1. Dredging/Sediment Removal
2. Aeration/Destratification
3. Flushing/Dilution
4. Nutrient Precipitation/Inactivation
5. Drawdown/Waterlevel Manipulation
6. Macrophyte Control
7. Biomanipulation
8. Sediment Sealing

Watershed Techniques

9. Agricultural BMP's
10. Stormwater Control
11. Erosion Control
12. Tributary Diversion/Treatment

Restoration Techniques

STATE and LAKE	1	2	3	4	5	6	7	8	9	10	11	12
California												
Ellis	x		x			x				x	x	
Gibraltar	x								x		x	
Lafayette	project terminated prior to restoration											
Stanford	x								x		x	
Temescal	x										x	
Connecticut												
Warramug										x	x	
Bantam	x									x	x	
Florida												
Apopka	project terminated prior to restoration											
Eola								x		x		
Jackson										x	x	
Illinois												
Frank Holten	x									x	x	
Indiana												
Skinner	x								x	x	x	

	Restoration Techniques											
STATE and LAKE	1	2	3	4	5	6	7	8	9	10	11	12
Iowa												
Blue	x											
Green Valley	x								x		x	
Lenox	x								x		x	
Manawa	x											
Olwein	x										x	
Kansas												
Lone Star	x										x	
Louisiana												
City	x								x	x	x	
Maine												
Cobbossee I		x		x					x		x	
Cobbossee II									x	x	x	
Little							x					
Sabasticook					x				x			
Sabattus					x				x			
Salmon									x	x		
Maryland												
Loch Raven		x										
Waterford										x	x	
Massachusetts												
Big Alum										x	x	x
Charles River		x										
Cochituate	x									x	x	
Ellis Brett	x									x	x	
Lower Mystic		x		x								
Morse	x			x		x			x		x	
Nutting	x								x	x	x	
Michigan												
Lansing	x								x		x	
Reeds		x								x		
Minnesota												
Albert Lea	x				x				x	x	x	x
Clear	x									x	x	
Hyland		x			x				x	x	x	
Long Lake Chain	x								x	x	x	
Penn	x	x								x		
Phalen										x		
Moore	x			x						x	x	
Clearwater River Chain				x						x		x
Harriet/Isles										x		
Missouri												
Creve Coeur	x								x			
Finger			x									x
Rothwell	x						x					
Vandalia	x						x					
Montana												
Sacajawea	x											x
Nevada												
Tahoe I										x	x	
Tahoe II										x	x	
New Jersey												
Allentown	x									x	x	
New York												
Ann Lee	x									x	x	
Buckingham	x				x							
Collins	x									x	x	
Delaware Park	x	x								x		x
Fifty-ninth Street	x							x			x	
Hampton Manor	x				x					x	x	
Hyde Park	x									x	x	
Ronkonkoma	x									x	x	
Scudders	x									x	x	
Steinmetz	x				x				x	x	x	
Tivoli					x					x	x	
Washington Park	x											
North Carolina												
Mystic	x										x	

	Restoration Techniques											
STATE and LAKE	1	2	3	4	5	6	7	8	9	10	11	12
Ohio												
Summit		x								x		x
Oklahoma												
Pauls Valley	x				x				x		x	
Sunset	x				x				x	x	x	
Oregon												
Commonwealth	x		x		x				x		x	
Pennsylvania												
North Park	x									x	x	
South Carolina												
Broadway	x									x	x	
South Dakota												
Cochrane						x				x	x	
Covell	x									x	x	
Herman	x								x		x	
Kampeska										x	x	
Oakwood											x	
Swan									x		x	
Sylvan	x								x	x	x	
Texas												
McQueeney	x									x		
Vermont												
Bomoseen						x						
Virginia												
Rivanna		x							x	x		
Washington												
Ballinger		x			x						x	x
Fenwick										x	x	x
Liberty	x			x					x	x		
Long	x			x						x	x	
Medical			x									x
Moses	x		x									
Sacajawea			x							x		
Spada/Chaplain			x						x		x	
Thurston									x	x	x	
Vancouver			x							x		
Wapato			x		x	x				x		
Wisconsin												
Bugle	x										x	
Decorah	x								x			
Half Moon	x									x		
Henry	x								x		x	
Lilly	x			x								
Little Muskego	x			x								
Marinuka	x										x	
Mirror/Shadow		x		x						x		
Noquebay					x	x			x	x		
Upper Willow	x											
White Clay									x			x

Appendix B

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PLENARY SESSIONS

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WORKING SESSIONS B

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